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CONTENTS

ARTICLES

- Diets of Age-0 Striped Bass in the San Francisco Estuary, 1973-2002 Marade E. Bryant and Jane D. Arnold 1
- The Ecology of Over-Water Nesting Ducks in Northeastern California.....Sabrina M. Wright-Myers and Raymond J. Bogiatto 23
- Feeding Preferences and Size-Related Dietary Shifts of Treefish (Scorpaenidae: *Sebastes serriceps*) off Southern California Ellen T. Kosman, Madhavi A. Colton, and Ralph J. Larson 40

NOTES

- Consideration of the Common Name for Gila (*Siphateles*) *bicolor vaccaceps*, a Tui Chub in the Cow Head Basin of Northeastern California Stewart B. Reid 49
- First Record of the Wolf-Eel, *Anarrhichthys ocellatus* (Pisces: Anarrhichadidae), from Baja California, Mexico Richard F. Feeney, Robert N. Lea, Sean Dyer, and Scott Gietler 52

DIETS OF AGE-0 STRIPED BASS IN THE SAN FRANCISCO ESTUARY, 1973-2002

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Age-0 striped bass, *Morone saxatilis*, abundance indices from the Summer Townet Survey in the San Francisco Estuary have declined since the 1970s and changes in potential prey composition have been thought a possible cause. Nearly 31,900 age-0 striped bass stomachs were collected between 1973 and 2002 and subsequently analyzed to monitor diet and evidence of feeding difficulties. Age-0 striped bass include many introduced zooplankton species in their diet. The percent of stomachs containing food have remained consistent over the past 30 years and the area west of the confluence of the Sacramento and San Joaquin rivers had the highest percent of stomachs containing food. There has been no apparent trend in the mean estimated weight of stomach contents, although striped bass have shifted from consumption of native zooplankton to consuming primarily introduced zooplankton. Three introduced copepod species (*Pseudodiaptomus forbesi*, *Pseudodiaptomus marinus*, and *Sinocalanus doerrii*) now compose the majority items eaten by age-0 striped bass, replacing the native copepod, *Eurytemora affinis*. The native mysid, *Neomysis mercedis*, has been replaced by introduced *Acanthomysis* spp. as the dominant mysid in age-0 striped bass diets. The results of this study indicate age-0 striped bass caught by the Summer Townet Survey have adjusted to changes in the summer food web and they do not appear to be starving as a result of changes in prey species or their numbers during this period. Impact, if any, occurred at the time of the larvae's critical first feeding, an earlier life stage than fish caught by the TNS.

INTRODUCTION

Striped bass, *Morone saxatilis*, were first introduced to the Pacific coast in 1879, when about 135 fish from New Jersey were planted in the San Francisco Estuary (Estuary). An additional plant of about 300 fish occurred in 1882; and by 1888, a commercial fishery had developed (Skinner 1962). The commercial fishery closed in 1935, and the fishery has since been reserved exclusively for sport anglers (Skinner 1962). Striped bass are widespread along the Pacific coast, but they are much less abundant than they were during the first 75 years or so following their explosive invasion of the region (Moyle 2002).

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In the 1970's, both age-0 and adult striped bass abundances began declining (Stevens et al. 1985). The CDFG Summer Townet Survey (TNS) 38.1-mm striped bass indices have declined from a historic high of 117.2 in 1965, to 0.9 in 2005 (CDFG, unpublished data). Adult striped bass abundance ranged from 1.5 to 1.9 million between 1969 and 1976, and decreased to only 579,000 in 1994 (Kohlhorst 1999). The striped bass decline has been attributed to reduced food supply, toxic herbicide contaminants, reduced egg production, and entrainment of young life stages on water diversion screens which cause reduced recruitment to the adult stock (Stevens et al. 1985; DFG 1992).

In other estuaries, lack of food and feeding habits were prominent among hypotheses for poor age-0 striped bass abundance (Martin et al. 1985, Chesney 1989, Cooper et al. 1998). Prey density is one of the important factors affecting age-0 striped bass survival and growth, and causing variability of the recruitment success of striped bass (Tsai 1991). The presence of adequate densities of suitable food organisms is of crucial importance to the survival and growth of larval white sucker, pumpkinseed, and herring and for recruitment success (Werner and Blaxter 1980, Hart and Werner 1987). An inadequate food supply is often a major factor in recruitment failure of several fishes (Houde 1978; Werner and Blaxter 1980). The elimination of a major food item during the critical period, when fish first start feeding, could rapidly increase mortality rates, decrease year class strength, and ultimately influence the condition of some centrarchids (Lemly and Dimmick 1982). The success of first-feeding larvae in the Estuary may be influenced by the recent invasion of exotic copepods that appear to be displacing native copepods previously important as food for striped bass (Meng and Orsi 1991).

Several species of zooplankton have been introduced to the Estuary over the past 30 years (Orsi et al. 1983, Orsi and Walter 1991), leading to substantial changes in the prey available to young fish. The first specimens of *Sinocalanus doerrii* were taken in May 1978 (Orsi et al. 1983). *Pseudodiaptomus marinus* was first observed in October 1986 (Orsi and Walter 1991) and *P. forbesi* first appeared in October 1987 (Orsi and Walter 1991). Two species of *Acanthomysis* began to appear in plankton samples collected since 1992, *A. aspera*, first caught August 1992, and *A. bowmani*, first caught July 1993 (Modlin and Orsi 1997). *Gammarus daiberi* were first detected in the Estuary in 1983, and has routinely been collected since 1986 (Hymanson et al. 1994).

These zooplanktons have the potential to affect age-0 striped bass diets since they may displace native species, may be more difficult to catch, or may not be abundant at the appropriate time and place for striped bass to encounter them. The increase in the numbers of exotic copepods happened simultaneously to the decline in abundance of the native copepod, *Eurytemora affinis*, the principal food of larval bass, although the decline was already under way by 1972 (Orsi and Mecum 1986). Exotic species may not be readily available to striped bass larvae. *S. doerrii* tends to be found at lower salinities than *E. affinis* (Orsi et al. 1983), and its longer antennae (used to assist swimming) may better enable it to escape predation. *P. forbesi* is morphologically similar to *E. affinis*, but it seems to prefer warmer water, and it becomes abundant later in the year after larval fish numbers have peaked (CDFG, unpublished data).

Previous studies conducted on the diets of age-0 striped bass in the Estuary

(Heubach et al. 1963 and Thomas 1967) were of short duration and occurred prior to changes in the food web. Thomas (1967) studied juvenile and adult striped bass, providing little data on age-0 striped bass food habits. Of the age-0 striped bass analyzed, he stated they rarely eat fish, but ate invertebrates almost exclusively. His study did not identify which invertebrates were consumed. The study by Heubach et al. (1963) was conducted year-round on age-0 striped bass diets and the results divided into geographic areas. The study by Heubach et al. (1963) showed the importance of *Neomysis*, especially since they are larger than copepods, so their importance would be greater on a weight basis. The importance of copepods and cladocerans to bass less than 1 inch was stated as well.

The TNS has been used to collect 30 years of age-0 striped bass diet data, enabling an analysis of the long term trends in the diet of striped bass over a period of change in prey composition. We examined age-0 striped bass diet data from 1973 to 2002, looking for a link between changes in prey species and feeding success. We ask the questions if striped bass are able to utilize these introduced species and if there is a link between changes in the food web and the abundance indices of striped bass. These results may identify one of many possible causes for the decline in abundance indices and show us further diet studies for earlier life stages of striped bass than those caught by the TNS may be beneficial.

STUDY AREA

The rearing area for the largest population of striped bass in California is concentrated in the San Francisco Estuary, which includes San Francisco Bay, San Pablo Bay, Suisun Bay, and upstream into the Sacramento and San Joaquin rivers and Delta (Fig. 1). Tidal movement dominates the Estuary, with flow reversal occurring in summer up to Courtland on the Sacramento River and above Mossdale on the San Joaquin River (Chadwick 1964). The Delta was once seasonally brackish, but is now maintained as essentially fresh water, with a salinity gradient toward the lower bays.

In the summer age-0 bass are concentrated primarily in the central-to-upper Estuary as delineated by the station locations. This area is bound roughly by the cities of Sacramento to the north, Stockton to the east, Tracy to the south, and by San Pablo Bay to the west (Heubach et al. 1963).

METHODS

Age-0 striped bass collected in the TNS from 1973 to 2002 were used in these analyses. The TNS sampled 32 stations every other week from eastern San Pablo Bay to Rio Vista on the Sacramento River and to Stockton on the San Joaquin River. The survey was scheduled for the 5-day time period of least tidal fluctuation, so any correlation between catch and tidal stage would not bias the results (Chadwick 1964). The TNS usually began the 3rd or 4th week of June, and continued until late July or early August, when the mean size of striped bass surpassed 38.1-mm (*i.e.* the size used to set the annual abundance index).

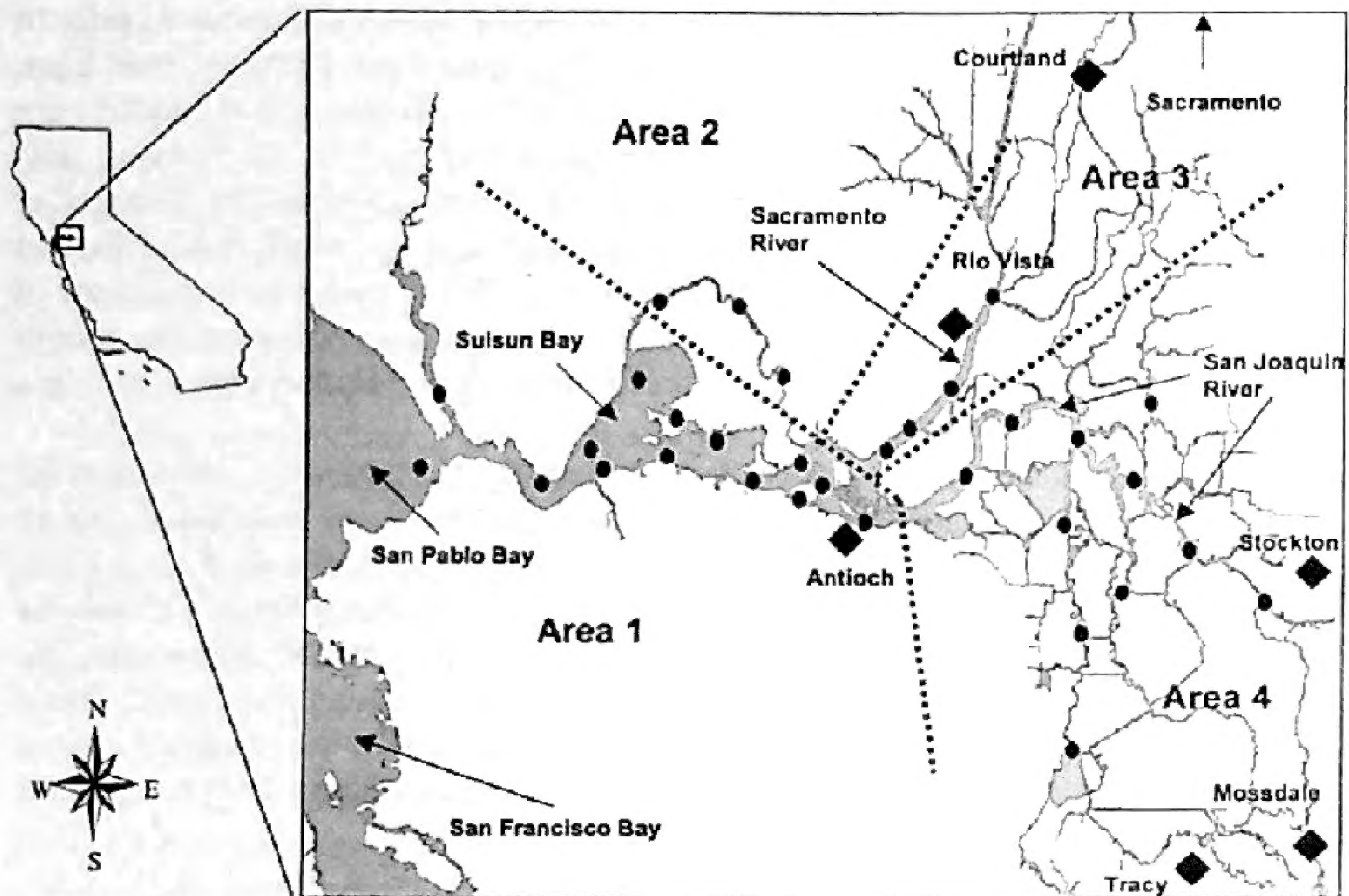


Figure 1. Map of Townet Survey stations in the San Francisco Estuary.

The gear used was a 12.7-mm ($\frac{1}{2}$ inch) stretched mesh nylon conical net with a 1.5 m² mouth opening lashed to a hoop frame and mounted on skis (Calhoun 1953). Three, 10-minute oblique tows were made against the current at each station. The gear was most efficient for striped bass between 20 and 50-mm (Chadwick 1964), at which point the year class strength was believed to be set (Turner and Chadwick 1972).

In the field, striped bass were put in perforated Ziploc bags and preserved in gallon jugs with 5% formalin. Fish were measured to the nearest millimeter fork-length in the lab prior to stomach analysis and divided into three size groups: 1) ≤ 25 mm, 2) 26 to 49 mm, and 3) ≥ 50 mm. The fish were categorized into four geographical areas: 1) Suisun Bay, Honker Bay, San Pablo Bay and Napa River, 2) Montezuma Slough and Grizzly Bay, 3) Sacramento River, and 4) the east and south delta.

The process for stomach content analysis began with fish being placed on their right sides and their left sides were cut open to remove the esophagus and stomach. For bass ≥ 15 -mm, the intestine was also removed, dissected, and examined. The presence or absence of food was recorded, and the food organisms present were identified to the lowest feasible taxon. To minimize the possibility of counting the same food organism more than once, whole organisms or heads only were counted. If other body parts were identifiable as distinctly different organisms, they were also counted.

The methodology used to collect diet data varied through time. For years 1973 to 1986, stomach contents were combined for each of the three size groups to determine diet composition and zooplankton species were lumped into one of five taxonomic

categories (copepods, cladocerans, amphipods, mysids, and other), with organisms of interest further differentiated. For years 1987 to 2002, the taxonomic levels became more specific and food organisms were identified and recorded for individual fish.

To examine the relationships between fish length and prey length, length measurements were taken only for a subset of the larger food items observed from 1993 to 2002: *Mysid* spp., *Crangon* spp., *Palaemon* spp., *Gammarus* spp., and *Corophium* spp. Invertebrate prey items were measured from the head to the tip of the telson to the nearest 0.5-mm. Larval fish were also measured to the nearest 0.5-mm.

Studies reporting on diet composition often report percent diet composition by number, weight (or volume) and frequency of occurrence (*i.e.* the percent of stomachs with food that contain a particular prey type) and often provide an index of relative importance incorporating all three. Each measure of diet composition has strengths and weaknesses (Hyslop 1980). Percent composition by number identifies numerically important diet items, but may overemphasize numerous small prey items of limited nutritional value. Percent composition by weight quantifies food types in direct comparable weight units, but may overemphasize rarely eaten large prey items that may have less digestible material. Percent composition by frequency of occurrence measures how wide-spread a particular prey item is in the diets of the sample population, but it does not reflect the importance of common items.

In addition to presenting amount of stomachs with food, diet data for 1973-2002 were summarized three ways. First, percent composition was calculated as the sum of organisms from each species or taxonomic group divided by the total number of organisms present in the stomach and multiplied by 100. We present these data by prey category to depict shifting trends across categories and by genus within categories to depict changes in prey species in diets. Second, frequency of occurrence was for a more limited period, 1987 to 2002, when diet was recorded for individual fish rather than for fish pooled by size groups. Lastly, mean weight of organisms consumed by age-0 striped bass was calculated as the number of organisms in striped bass stomachs multiplied by an estimated dry weight of the particular zooplankton species, divided by the total number of striped bass. This provides an estimate of an average ration and its composition. Only food items with weight data or estimates were used in the mean weight analysis (Table 1). For all copepods and cladocerans, only one weight was used regardless of the size of striped bass. For amphipods and mysids, a mean length by striped bass size group was determined using the length data from this study. The weights of food items were multiplied by the total number of each food item, summed by food group (copepod, cladoceran, amphipod, and mysid) and divided by the number of striped bass stomachs containing food to determine the mean weight of food by striped bass size group.

Diet data were graphed and examined qualitatively for time trends and abrupt changes.

RESULTS

We identified stomach contents of 31,823 age-0 striped bass collected from 1973 to 2002. An average of 74% of the stomachs contained food, with a range of 54% (1974)

Table 1. Weights in micrograms for food items used to calculate mean weight.

Taxa	Weight	Source
Cyclopoid	8	Culver et al. 1985
<i>Eurytemora affinis</i>	3.5	Cathy Hall, UCD, personal communication
<i>Sinocalanus doerrii</i>	9	CDFG Unpublished Data
<i>Pseudodiaptomus forbesi</i>	3.5	Cathy Hall, UCD, personal communication
Harpacticoid	2.8	CDFG Unpublished data
Other copepod	6	CDFG Unpublished data
<i>Oithona davisii</i> and <i>O. similis</i>	0.6	CDFG Unpublished data
<i>Bosmina longirostris</i>	1.5	Culver et al. 1985
<i>Daphnia</i> spp.	2.5	Dumont et al. 1975
<i>Diaphanasoma brachyurum</i>	1.5	Culver et al. 1985
Other cladocerans	2	CDFG Unpublished data
Mysids		
In bass \leq 25-mm	11.8	CDFG Unpublished data
In bass 26-49-mm	525	CDFG Unpublished data
In bass \geq 50-mm	900	CDFG Unpublished data

to 87% (1984), but there was no trend over time (Fig. 2). The highest average number of stomachs containing food was found in the Sacramento River (84.9%) and lowest west of the confluence of the Sacramento and San Joaquin Rivers (70.2%). The number of stomachs examined varied substantially from year to year, due to changes in catch attributable to year class size (Table 2).

Percent Composition by Number

Copepods were the dominant food organism for all size classes of age-0 striped bass (Fig. 3), and more were consumed after 1989 than prior to 1989. Striped bass < 50 mm consistently consumed more copepods than striped bass ≥ 50 mm. Copepods represented over half of the stomach contents of striped bass ≤ 25 mm in 80% of the years, and represented over half of stomach contents for striped bass ≥ 50 -mm in only 47% of the years (Fig. 3). The highest numbers of copepods were consumed west of the confluence of the Sacramento and San Joaquin Rivers (38.3%), and the lowest number was consumed in the Sacramento River (11.1%).

Eurytemora affinis, a preferred food item of age-0 striped bass (Meng and Orsi 1991) and a historically dominant species in the upper Estuary (Orsi and Mecum 1986), was assumed to be the predominant copepod in this category prior to 1980, when copepods were not identified to species. The majority of *E. affinis* were consumed in the south and east delta (52.8%). The introduction of *Sinocalanus doerrii* in 1979 necessitated more refined stomach content identification. The majority of *S. doerrii* were consumed in the east and south delta (53.2%). *Pseudodiaptomus* spp. appeared in age-0 striped bass stomachs in 1987, and became the dominant copepods after 1989 (Fig. 4). The

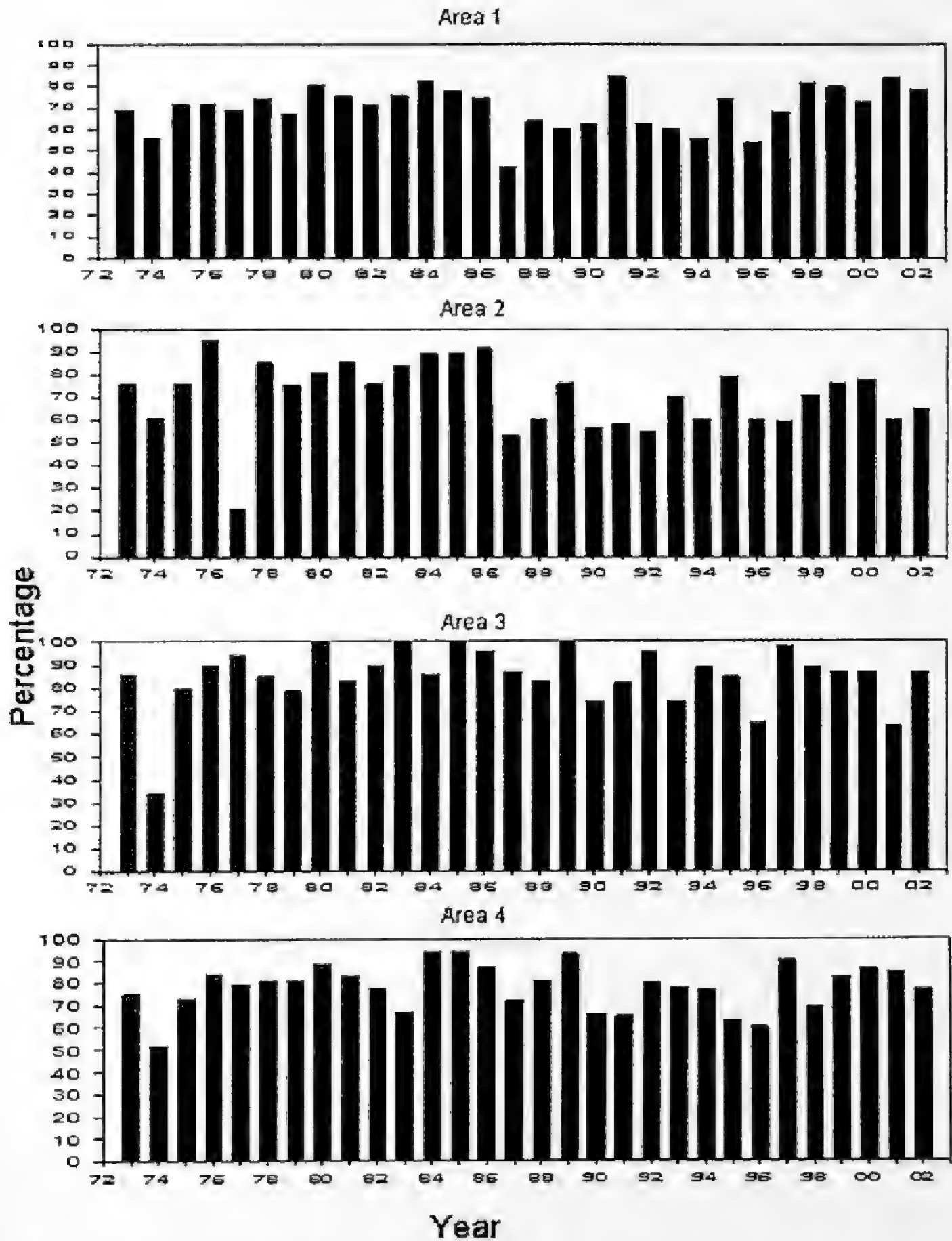


Figure 2. Percent of age-0 striped bass stomachs examined with food by area 1973-2002.

Table 2. Number of age-0 striped bass stomachs with food by year and area.

Year	Area 1	Area 2	Area 3	Area 4	Total with food	Total Examined
1973	507	273	205	534	1519	2047
1974	523	228	65	350	1166	2169
1975	882	376	377	573	2208	2967
1976	275	127	127	215	744	911
1977	120	9	117	143	389	523
1978	498	265	270	627	1660	2075
1979	208	220	70	223	721	968
1980	227	162	1	178	568	682
1981	240	127	165	212	744	922
1982	797	361	95	333	1586	2130
1983	349	368	2	29	748	945
1984	331	216	18	133	698	802
1985	187	278	24	109	598	694
1986	424	205	267	458	1354	1591
1987	152	146	189	214	701	1149
1988	134	35	204	259	632	831
1989	74	121	15	198	408	510
1990	104	124	57	87	372	598
1991	111	84	106	127	428	599
1992	227	135	149	305	816	1149
1993	444	206	92	282	1024	1519
1994	231	98	146	184	659	974
1995	376	331	29	91	827	1098
1996	30	106	11	9	156	265
1997	77	55	46	27	205	284
1998	278	320	17	55	670	888
1999	151	205	26	55	437	555
2000	197	287	162	128	774	974
2001	179	37	160	152	534	714
2002	22	70	27	96	215	290
Average Percent	70.2	70.6	84.9	78.1	74.0	
MIN	42.5	21.4	33.9	52.2	53.8	
MAX	85.4	95.5	100	94.3	87.0	
SE	1.88	2.78	2.47	1.89	1.45	
C.I. (95%)	3.85	5.68	5.1	3.86	2.97	

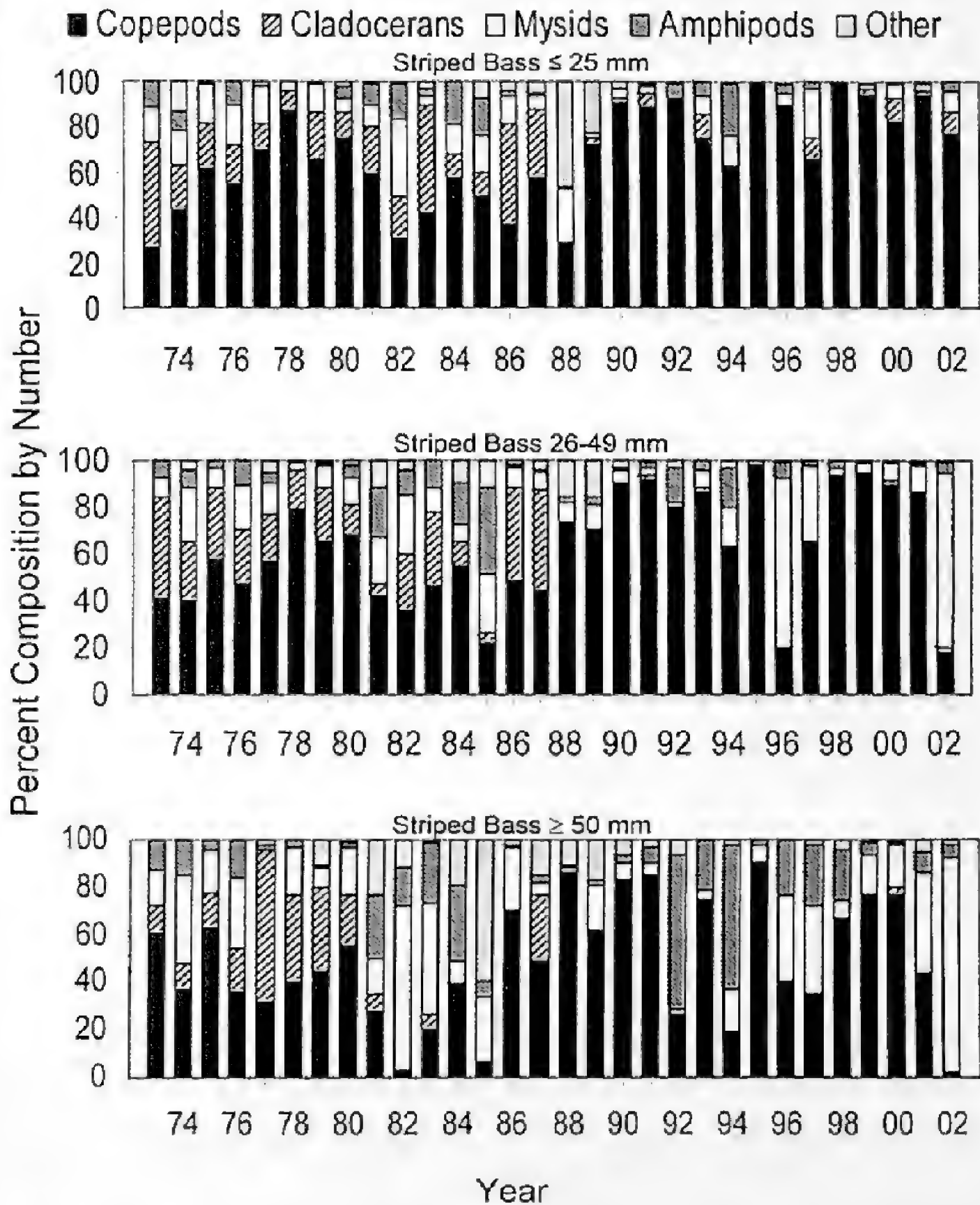


Figure 3. Percent composition by number of categorized diet items in stomachs of 3 size classes of age-0 striped bass 1973-2002.

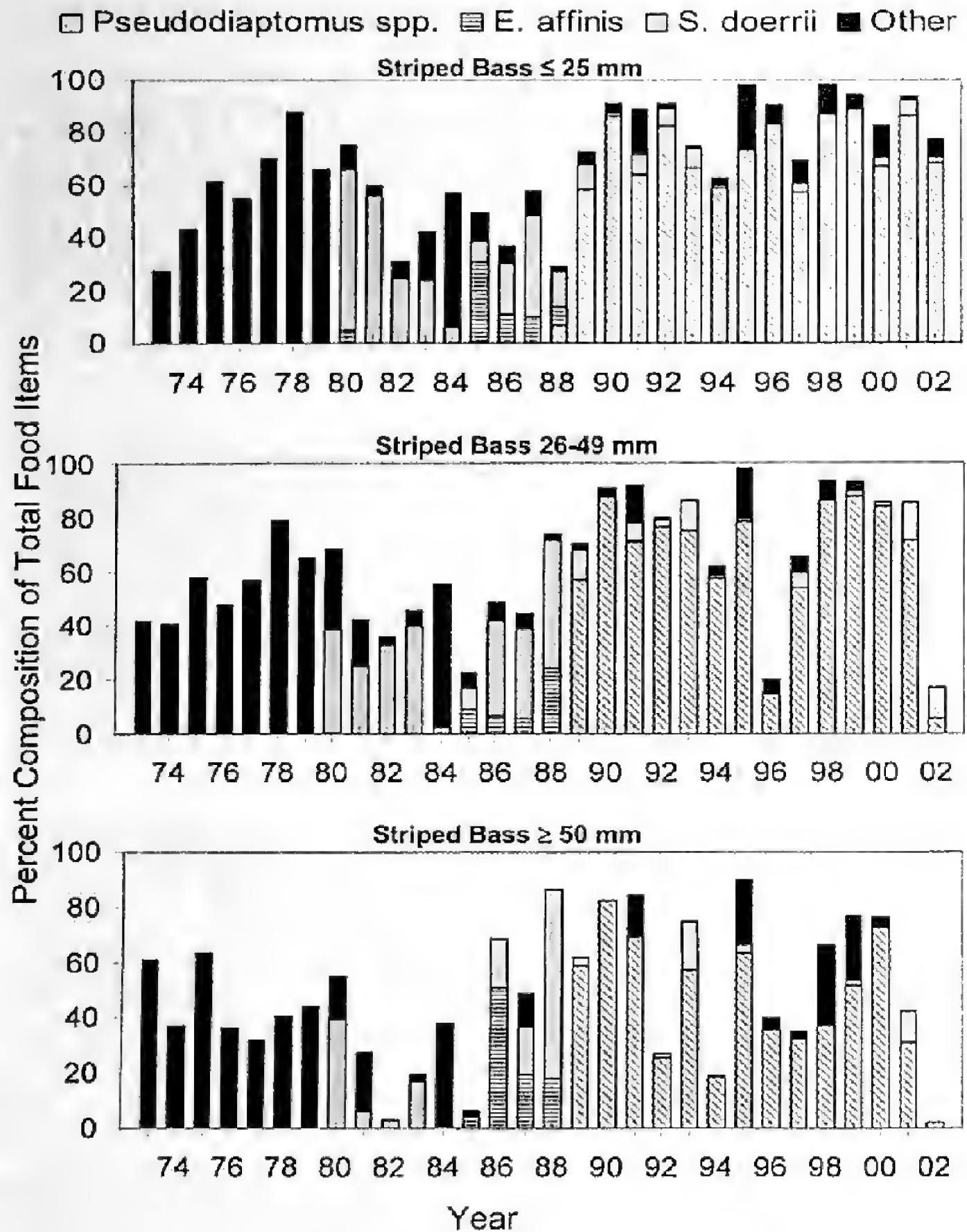


Figure 4. Copepods as a percent of the total items in stomachs of 3 size classes of age-0 striped bass 1973-2002.

majority of *Pseudodiaptomus* spp. were consumed west of the confluence of the Sacramento and San Joaquin Rivers (47.6%).

The presence of once common *Cladocera* spp. in all age-0 striped bass size classes have declined considerably since 1988 (Fig. 3). The average annual percent composition of cladocerans dropped from 21.0 from 1973-1987 to 1.6 from 1988 to 2002. After 1987, *Cladocera* spp. were uncommon (average of 3.8% of stomach contents) in striped bass ≤ 25 mm and almost non-existent (average of 0.6% of stomach contents) in striped bass > 25 mm. The highest percentage of cladocerans consumed was in the east and south delta (93.1%), and the lowest west of the confluence of the Sacramento and San Joaquin Rivers (1.5%).

Neomysis mercedis was a prominent food item for age-0 striped bass until the introduction of *Acanthomysis* spp. in 1992 (Fig. 5). *Acanthomysis* spp. was first recorded in striped bass stomachs in 1994, and has since become a major food item, especially with fish > 25 mm. The presence of *N. mercedis* in all size classes of age-0 striped bass diets has been virtually zero (average of 0.3% of stomach contents) since 1993. *Neomysis mercedis* and *Acanthomysis* spp. numerically represented a higher percentage of stomach contents in fish ≥ 50 mm (15.0%) compared with fish ≤ 25 mm (7.3%) (Fig. 5). The highest percentage of mysids was consumed west of the confluence of the Sacramento and San Joaquin Rivers (44.8%), and lowest in the east and south delta (6%).

Amphipods were a small portion (average of 5.9% of stomach contents) of age-0 striped bass diets for fish < 50 mm. Striped bass ≥ 50 mm generally consumed higher numbers of amphipods (average of 14.0% of stomach contents for all years), and even more in the early 1990s (average of 31.4% of stomach contents from 1990 to 1994) than any other time period of the study (Fig. 3). The highest percentage of amphipods was consumed in the east and south delta (50.1%), and the lowest in the Sacramento River (7.1%). *Corophium* spp. have declined in abundance in all size classes of age-0 striped bass stomachs from 9.4% of stomach contents from 1973 to 1984, to 2.9% of stomach contents since 1985 (Fig. 6). A brief increase in consumption occurred in 1996 (23.4% of stomach contents) among striped bass ≥ 50 mm, but have since dropped to 3.4% of stomach contents in the following 6 years. *Gammarus* spp. first appeared in age-0 striped bass stomachs in 1987, but contributed little until the early 1990s (Fig. 6). Striped bass ≥ 50 mm saw a larger contribution to the percent of stomach contents (average of 15.5%) than striped bass < 50 mm (average of 3.0%).

Except in the mid-late 1980s, organisms categorized as "Other" (larval fish, shrimp, and insects) comprised very little of the diet of age-0 striped bass (Fig. 3).

Frequency of Occurrence

From 1987-2002, when contents of individual stomachs were recorded, patterns in frequency of occurrence closely approximated those of percent composition by number for striped bass ≤ 25 mm, indicating percent composition by number was not driven by limited numbers of fish ingesting high numbers of specific organisms.

Copepods had the highest annual frequency of occurrence (average of 74.5 per

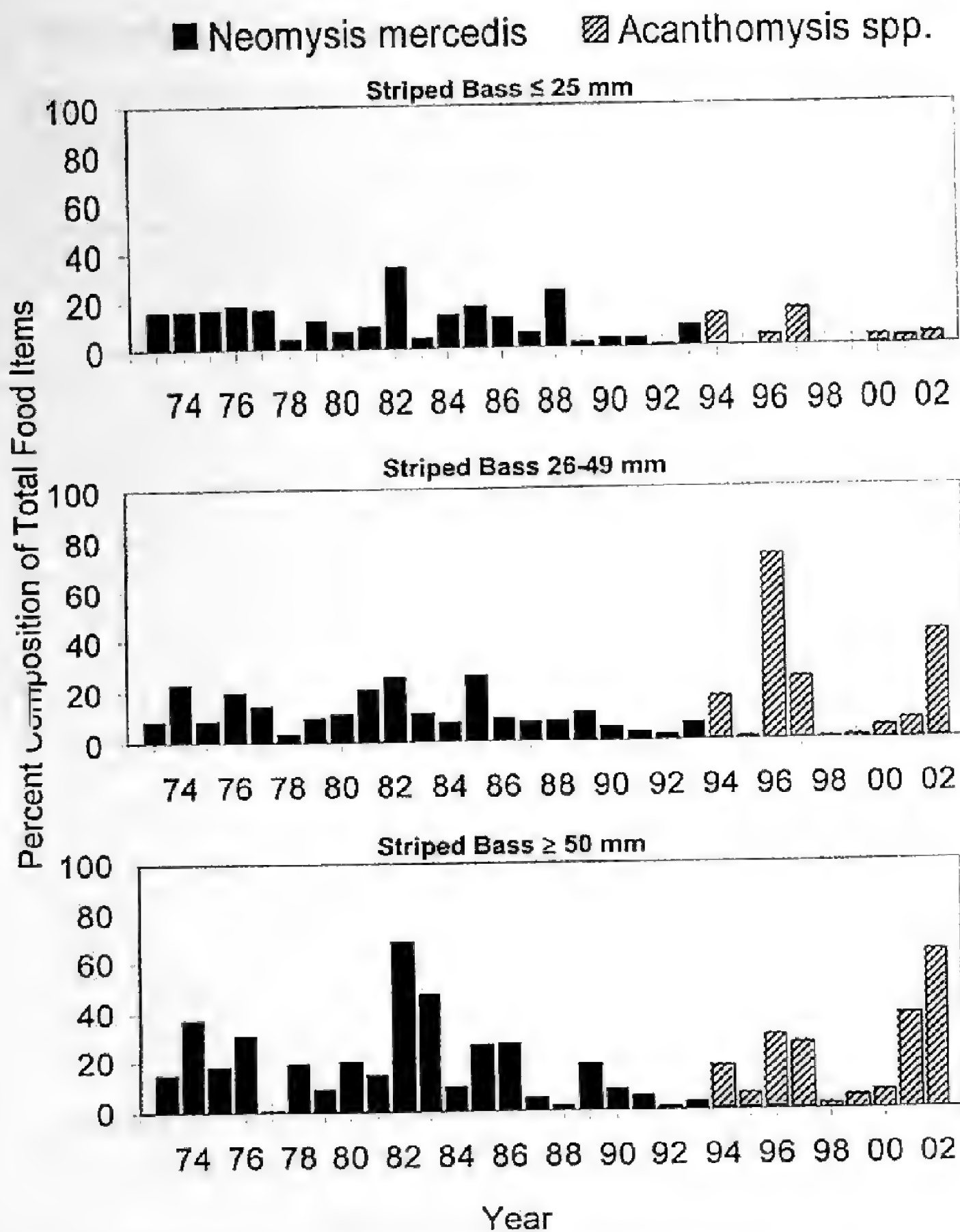


Figure 5. *Neomysis mercedis* and *Acanthomysis* spp. as a percent of total items in age-0 striped bass stomachs 1973-2002.

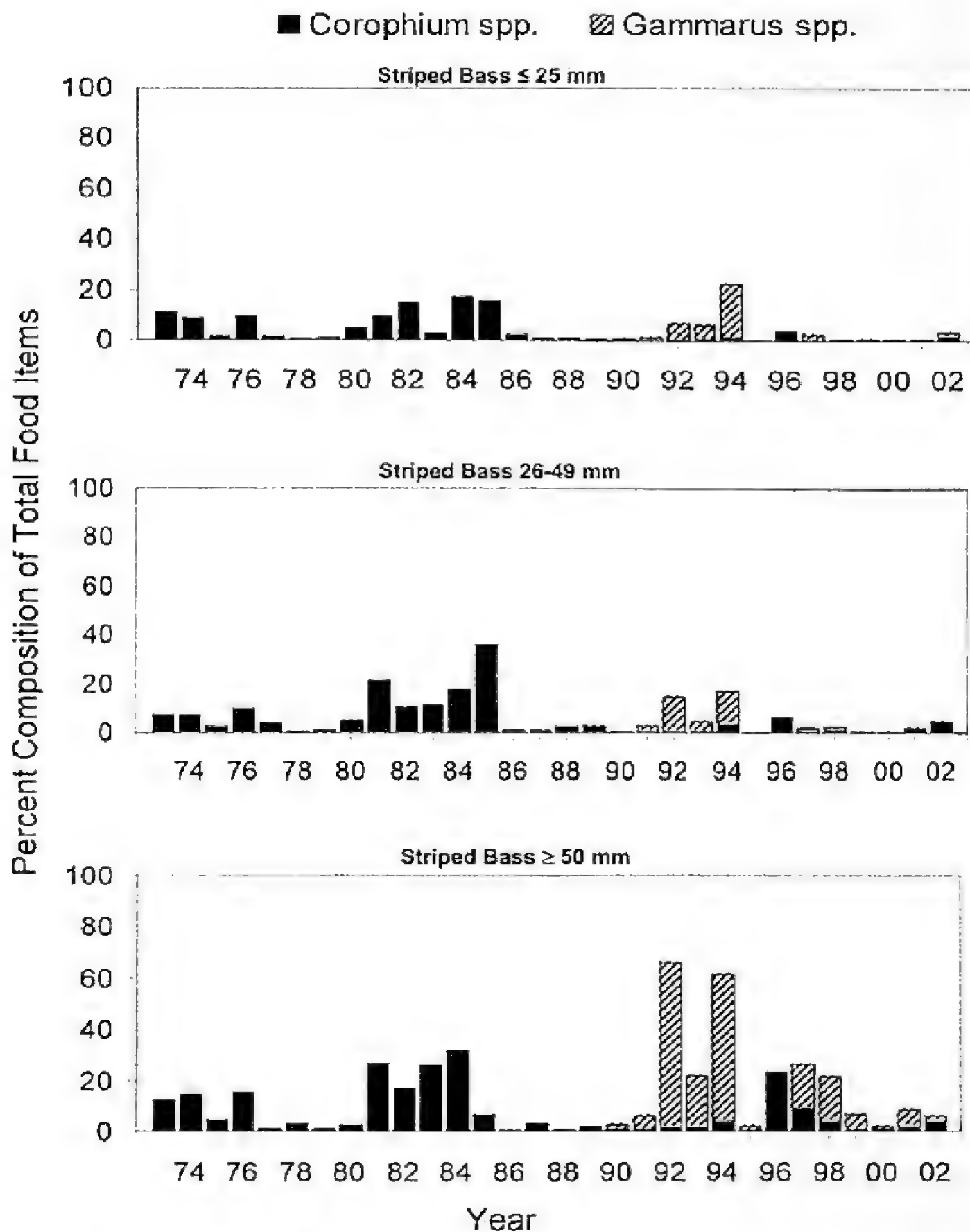


Figure 6. *Corophium* spp. and *Gammarus* spp. as a percent of total items in age-0 striped bass stomachs 1973-2002.

stomach) in striped bass ≤ 25 mm (Fig. 7). This was compared with the frequency of occurrence for copepods (average of 24.4 per stomach) in striped bass ≥ 50 mm, and with the frequency of occurrence of all other groups of organisms (average per stomach of 11.0 for cladocerans, 22.1 for mysids, and 22.2 for amphipods) in striped bass ≤ 25 mm.

Cladocerans were never found in a high proportion of striped bass stomachs of any size class. Frequency of occurrence was highest ($\leq 30\%$) in striped bass ≤ 25 mm (some were consumed in all years except one), and declined in larger size classes. Striped bass ≥ 50 mm had 7 years with cladocerans present in the stomachs and the frequency of occurrence averaged 3.8 individuals per fish of the fish that consumed cladocerans (Fig. 7).

All size classes of age-0 striped bass consistently ate mysids and amphipods (Fig. 7). Striped bass ≤ 25 mm averaged 22.1 mysids per stomach, striped bass 26-49 mm averaged 46.0, and striped bass ≥ 50 mm averaged 46.3. Striped bass ≤ 25 mm averaged 22.2 amphipods per stomach, striped bass 26-49 mm averaged 33.0, and striped bass ≥ 50 mm averaged 50.7. Striped bass > 25 mm had higher frequency of occurrences of amphipods (average of 41.8 per stomach) and mysids (average of 46.1 per stomach) than striped bass ≤ 25 mm, but striped bass ≤ 25 mm ate more amphipods and mysids than cladocerans.

Age-0 striped bass diets showed a decline in the frequency of occurrence of amphipods eaten after 1994 in all size classes. From 1987 to 1994, striped bass ≤ 25 mm had an average frequency of occurrence of 34.2 amphipods per stomach, and dropped to an average of 10.2 between 1995 and 2002. Striped bass 26-49 mm had an average frequency of occurrence of 43.7 amphipods per stomach between 1987 and 1994, and dropped to an average of 22.2 amphipods per stomach after 1994. Striped bass ≥ 50 mm had an average frequency of occurrence of 60.5 amphipods per stomach between 1987 and 2002, and dropped to an average of 40.9 amphipods per stomach after 1994. All three size classes showed an average decline of approximately 20 amphipods per stomach after 1994.

Contribution by Mean Weight of Food Organisms

For the invertebrate diet items measured, the mean weight of food in age-0 striped bass stomachs increased in direct proportion to fish size (Fig. 8). As bass grew, they increasingly consumed larger organisms, such as mysids and amphipods, and ate fewer smaller copepods (Table 3). Copepods (average weight of 41.8 μg per stomach) and amphipods (average weight of 92.6 μg per stomach) contributed most to the mean weights of food items for striped bass ≤ 25 mm. Mysids and amphipods dominated by weight in the diet of striped bass > 25 mm (Fig. 8).

Mysids had the highest contribution of weight per stomach for striped bass > 25 -mm (average mysid weight per stomach was 2,472.3 μg). The weight contribution of mysids consumed in all size classes declined in the late 1980's, but resurged around 1996. The average weight of mysids in striped bass 26-49 mm was more than double

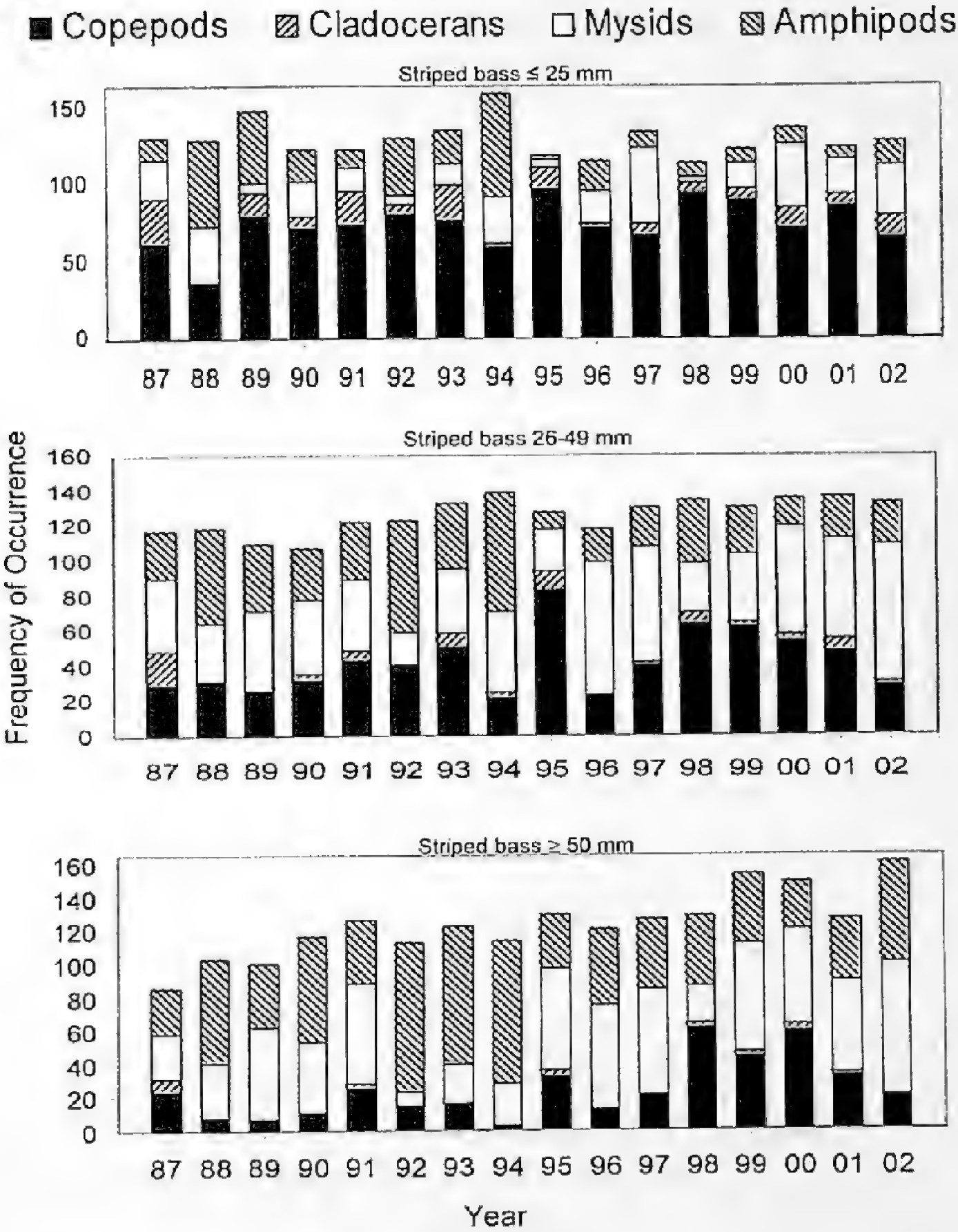


Figure 7. Frequency of occurrence of copepods, cladocerans, mysids, and amphipods in age-0 striped bass 1987-2002.

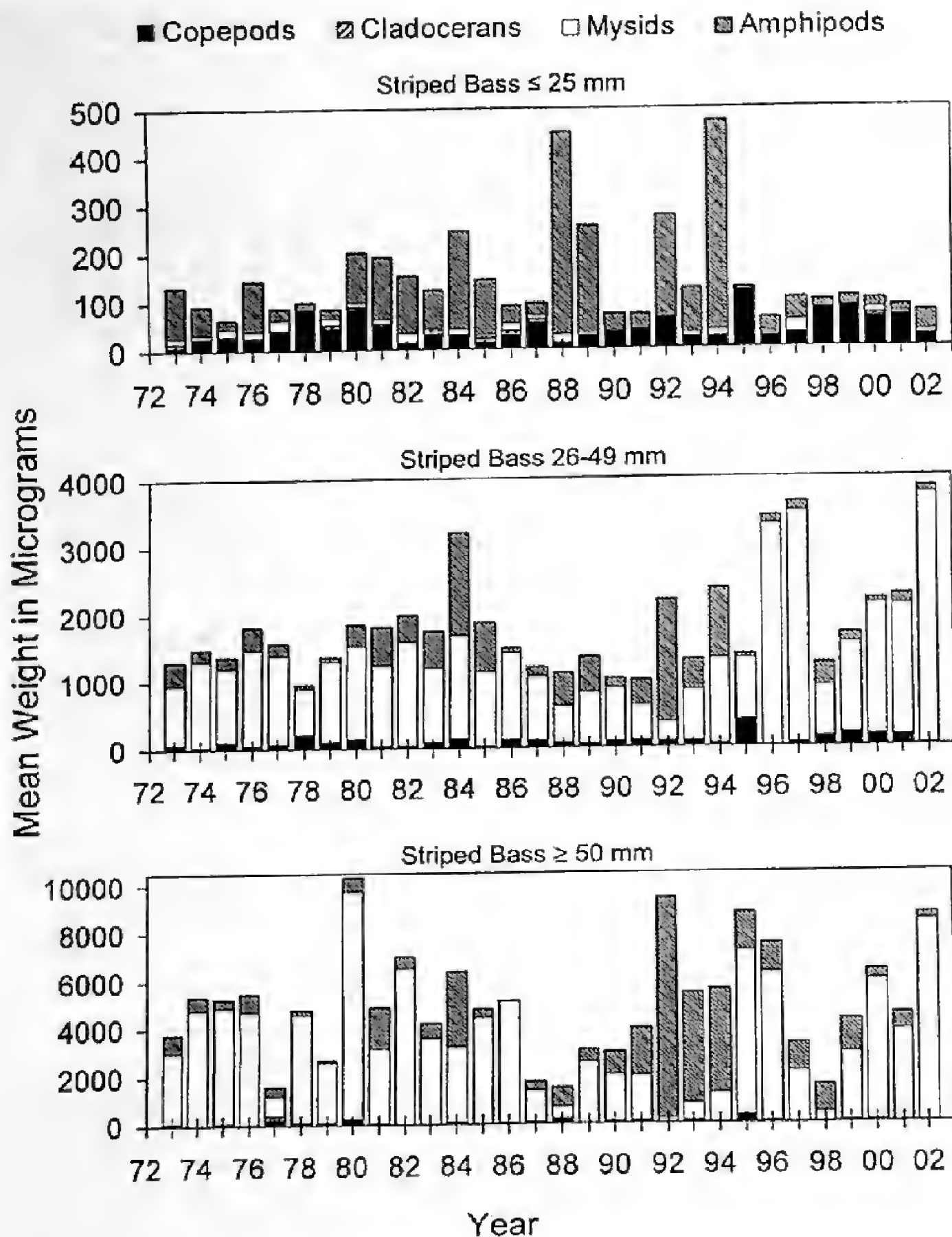


Figure 8. Mean weight of food items in age-0 striped bass 1973-2002.

Table 3. Invertebrate introductions into the San Francisco Estuary and dates of first identification in age-0 striped bass stomachs from the TNS.

Species	Year introduced	Year identified in stomachs
<i>Sinocalanus doerrii</i>	1978	1980
<i>Gammarus daiberi</i>	1983	1987
<i>Pseudodiaptomus marinus</i>	1986	1987
<i>Pseudodiaptomus forbesi</i>	1987	1987
<i>Acanthomysis aspera</i>	1992	1994
<i>Acanthomysis bowmani</i>	1993	1994

after 1996 (2359.4 μg) than the average weights per stomach of the previous 23 years (1,043.7 μg). Mysids had the highest average mean weight (1598.2 μg) in the Sacramento River and lowest (193.6 μg) in the east and south delta.

Amphipods had the second highest mean weights per stomach of age-0 striped bass > 25 mm, but for striped bass ≤ 25 mm they represented the organisms of highest weight per stomach. All three size classes of age-0 striped bass showed a decline in the average weight of amphipods per stomach beginning in 1995. Striped bass ≤ 25 mm went from averaging 117.4 μg per stomach to an average of 24.5 μg , striped bass 26–49 mm went from averaging 466.2 μg to 134.6 μg per stomach, and striped bass ≥ 50 mm went from averaging 1,490.6 μg to 976 μg per stomach. The highest average mean weight (567.1 μg) of amphipods occurred in the east and south delta and lowest (109.2 μg) in Montezuma Slough and Grizzly Bay.

The weight of cladocerans consumed was the lowest of any food organism for all size classes of striped bass (average weight of 2.9 μg per stomach for striped bass ≤ 25 mm, 5.9 mg for striped bass 26–49 mm, and 12 μg for striped bass ≥ 50 mm). The highest mean weight (13.9 μg) of cladocerans occurred in the east and south delta and lowest (0.2 μg) west of the confluence of the Sacramento and San Joaquin Rivers.

Striped bass < 50 mm consumed lower mean weights of copepods in the 1980s (average weight of 53 μg per fish) than in the 1970s (average weight of 57.2 μg per fish) and 1990s (average weight of 83.3 μg per fish). The highest average mean weight (59.8 μg) of copepods occurred in the east and south delta and the lowest (26.4 μg) in Montezuma Slough and Grizzly Bay. *E. affinis* and *S. doerrii* matched those results from the overall copepods, but *Pseudodiaptomus* spp. had the highest average mean weight (75.3 μg) west of the confluence of the Sacramento and San Joaquin Rivers and the lowest (27.8 μg) in Montezuma Slough.

DISCUSSION

Striped bass in marine or brackish water areas are generally considered to be opportunistic or nonselective feeders, consuming prey in accordance with the local community structure (Cooper et al. 1998). The diet of age-0 striped bass in the Estuary changed over the past 30 years with the declining abundances of *E. affinis*, *N. mercedis*,

and cladocerans (Orsi and Mecum 1986), and with the introductions of several species of zooplankton (Orsi et al. 1983; Orsi and Walter 1991). As the food web in the Estuary changed, age-0 striped bass from the TNS adapted to these changes by including new species of zooplankton into their diets, amid the decline of several native zooplankton species.

The decline in native zooplankton species showed up in age-0 striped bass diets from the TNS. Declines in all sizes of striped bass occurred in *N. mercedis*, *E. affinis*, *Cladoceran* spp., *Corophium* spp., and amphipods. A comparison with Heubach et al. (1963) shows the decline of several historic zooplankton species in the diets of age-0 striped bass. The frequency of occurrence of *N. mercedis* and *E. affinis* was substantially lower from 1987-2002 than reported in Heubach's 1956-1961 findings. Cladoceran consumption dropped significantly from 1987 to 2002, but they were significant food items in Heubach's study from 1956-1961. *Corophium* spp. all but disappeared in 1987-2002, while in 1956-1961 they were the third most consumed organism behind *N. mercedis* and copepods at year-round stations in the San Joaquin River. Heubach et al. (1963) found the occurrence of plankton species in juvenile striped bass stomachs generally reflected plankton abundances in the environment. The changes in the food web environment are reflected in the diets of age-0 striped bass from the TNS.

Boynton et al. (1981) stated it appeared juvenile striped bass feed primarily on those items most abundant in the water column and sediment surface. *Sinocalanus doerrii*, *P. forbesi*, and *Acanthomysis* spp. appeared in age-0 striped bass stomachs within a year or two after their introduction (Table 3). The lag can probably be attributed to the time it took the populations to build up to sufficient numbers in the environment to be readily available to striped bass (Orsi and Walter 1991). A gradual shift in consumption occurred from *E. affinis* to *S. doerrii* to *Pseudodiaptomus* spp. In the Delta, *E. affinis* dominated the fauna in all years before *S. doerrii* became established. *Sinocalanus doerrii* apparently displaced *E. affinis* downstream in 1979, and has dominated the Delta fauna in all years from then to 1985 (Orsi 1987). *P. forbesi* was the most abundant calanoid in Suisun Bay and Delta during the fall of 1988 and 1989 (Orsi and Walter 1991).

Salinity has been suggested as a major factor affecting the diets of juvenile striped bass (Heubach et al. 1963; Markle and Grant 1970), presumably by affecting the assemblage of prey items of sizes appropriate for juvenile striped bass (Boynton et al. 1981). Heubach et al. (1963) found the influence of salinity on the relative abundance of copepod genera in the diet of bass was readily recognizable. At stations with a high mean salinity, *Eurytemora* was the dominant genus, and as the salinity decreased upstream, percentages of freshwater genera increased (Heubach et al. 1963).

In the Estuary, *S. doerrii* may occupy a previously vacant niche, as copepods have never been abundant in its fresh water reaches, except in the San Joaquin River near Stockton and in adjacent sloughs (Orsi et al. 1983). Striped bass now have a supplemental food source which is present much farther upstream than *E. affinis*, which is most abundant in the entrapment zone (Orsi 1987). Since the preferred salinity ranges of *E. affinis* and *S. doerrii* differ, competition between the two is likely to be minimal (Orsi et al. 1983).

Striped bass preferred *P. forbesi* over *S. doerrii* (Meng and Orsi 1991), explaining why the majority of the copepods consumed after 1988 were *P. forbesi*. The larval fish studied by Meng and Orsi (1991) did not discriminate strongly between *E. affinis* and *P. forbesi*, indicating that *P. forbesi* may be a suitable food for young striped bass. Apparently, the gliding movement broken by short darts makes *E. affinis* and *P. forbesi* more vulnerable to predation than the jumping behavior of *S. doerrii* (Meng and Orsi 1991). Another important factor is the amount of time each species spends in motion, which increases prey visibility and reactive distance (O'Brien 1979). The hanging behavior of *S. doerrii* may make it more difficult to detect (Meng and Orsi 1991).

Even if *P. forbesi* is a suitable food organism for young bass, it needs to be available at the right time and place for the bass to utilize them. The TNS saw the highest percentage of *Pseudodiaptomus* spp. west of the confluence of the Sacramento and San Joaquin rivers, and this was the area with the highest average percent of stomachs with food from the TNS. We can conclude *Pseudodiaptomus* spp. is a suitable food for age-0 striped bass caught by the TNS, but perhaps *Pseudodiaptomus* spp. is not a suitable food for critical first-feeding larvae since *P. forbesi* does not become abundant until late spring (CDFG, unpublished data). The abundance of *E. affinis* peaks at the height of the striped bass spawning season; *S. doerrii* is also abundant at this time, but are not eaten until the fish reach 9-10 mm (CDFG, unpublished data). When fish reach this length, the critical first-feeding is over. It appears the success of the first-feeding larvae and the resulting year-class strength depends on the co-occurrence of larval striped bass with *E. affinis* (Meng and Orsi 1991). The fish caught by the TNS are the survivors from the critical first-feeding period.

From critical first-feeding to 20-50 mm (the size range of the highest gear efficiency from the TNS (CDFG, unpublished data), fish change their diets as they grow (Werner and Gilliam 1984). During development, it is common to find an increase in the size of food particles included in the diet (Siefert 1972, Wankowski 1979, Lemly and Dimmick 1982, Kane 1984, Mills et al. 1984, Peterson and Ausubel 1984, Govoni et al. 1986). Age-0 striped bass were no exception, consuming more amphipods and mysids than copepods as they grew. The average size of organisms consumed increased as age-0 striped bass size increased (Table 4). Heubach et al. (1963) found in general, the percentage frequency of copepod occurrence was greater in small bass than larger ones, while the frequency of occurrence of larger plankton, *Neomysis* and *Corophium*, was greater in larger bass.

Age-0 striped bass caught by the TNS in the Estuary are eating as much as they did 30 years ago, but what they are consuming has changed significantly due to zooplankton availability. We did not see a downward trend in the number of stomachs containing food or in the mean weights of organisms consumed, even in the 1970s when age-0 striped bass abundance indices began declining. Areas where age-0 striped bass consumed the most food were fitting with the literature as far as where particular organisms could be found and the sizes of organism consumed in relation to the size of striped bass caught by the TNS matched previous diet studies of striped bass in the Estuary. Striped bass caught by the TNS and used for the annual abundance indices do not show signs of feeding distress which could relate to the decline in indices. No

Table 4. Average size (mm) of macro-zooplankton species consumed by age-0 striped bass in each of the 3 size groups.

Species	≤ 25 mm	26-49 mm	≥ 50 mm
<i>Acanthomysis</i> spp.	Mean size = 2.5 N = 405 SE = 0.06 Min = 1 Max = 9	Mean size = 3.8 N = 2087 SE = 0.04 Min = 1 Max = 31	Mean size = 5.6 N = 364 SE = 0.1 Min = 1.5 Max = 10
<i>Corophium</i> spp.	Mean size = 1.9 N = 59 SE = 0.14 Min = 1 Max = 5	Mean size = 2.8 N = 235 SE = 0.11 Min = 1 Max = 8	Mean size = 4.6 N = 158 SE = 0.16 Min = 1 Max = 11
<i>Gammarus</i> spp.	Mean size = 1.7 N = 652 SE = 0.02 Min = 1 Max = 6	Mean size = 3.1 N = 1097 SE = 0.06 Min = 1 Max = 16	Mean size = 5.7 N = 931 SE = 0.07 Min = 1 Max = 12
<i>Neomysis mercedis</i>	Mean size = 3.1 N = 194 SE = 0.08 Min = 1 Max = 8	Mean size = 5.2 N = 769 SE = 0.1 Min = 1.5 Max = 13	Mean size = 7.1 N = 117 SE = 0.3 Min = 2 Max = 12.5
Unidentified mysids	Mean size = 2.4 N = 4 SE = 0.4 Min = 1.5 Max = 3	Mean size = 2.8 N = 52 SE = 0.3 Min = 1 Max = 8.5	Mean size = 2.9 N = 9 SE = 0.6 Min = 1.5 Max = 6

measure of food quality was looked at, but our results showed it is possible the decline is occurring before the age-0 abundance indices are set. A study on the larval diets of striped bass may show inability for first-feeding striped bass to incorporate new zooplankton into their diets.

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THE ECOLOGY OF OVER-WATER NESTING DUCKS IN NORTHEASTERN CALIFORNIA

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This study tested hypotheses on the relationship between nest site characteristics and nest success in over-water nesting ducks at Eagle Lake, Lassen County, California. Nest site characteristics may influence nest success in ducks, although previous studies investigating this relationship have yielded mixed results. Four study sites were chosen based on the presence of nesting habitat. Over-water habitats at Eagle Lake consisted of stands of hardstem bulrush, *Scirpus acutus*, and Baltic rush, *Juncus balticus*, with small patches of broadleaf cattail, *Typha latifolia*. Nesting habitat and nest site characteristics were quantified, and nest success was estimated. Lake-wide apparent nest success estimates during 2000 and 2001 for ruddy ducks, *Oxyura jamaicensis*, redheads, *Aythya americana*, lesser scaup, *Aythya affinis*, and cinnamon teal, *Anas cyanoptera*, were found to be 26.5% and 37.5%, 50% and 20%, 33.3% and 20% and 0% and 66.7%, respectively. Mayfield 40% nest success estimates during 2000 and 2001 for ruddy ducks was found to be 9.6% and 31.5%; for redheads, 16.6% and 5.8%; for lesser scaup, 3.9% and 45%; and for cinnamon teal, 5.3% and 63.4%. Water depth did not differ between successful and failed nests of ruddy ducks, scaup, and teal (ANOVA, all $P > 0.05$). However, in 2001, successful redhead nests were found in significantly deeper water than failed nests (ANOVA, $P < 0.05$). Using multiple regression analysis it was determined that nest visibility, percent cover, brood parasitism, and height of nesting cover did not significantly affect over-water nest success of ducks at Eagle Lake (all $P > 0.05$).

INTRODUCTION

As compared to dabbling ducks (Anatini), studies of wetland use by breeding diving ducks are relatively few. Consequently, multi-species studies of over-water nest

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habitat utilization are limited (Featherstone² 1975, Krasowski and Nudds 1986, Maxson and Riggs 1996).

Most studies of the nesting ecology of waterfowl in northern California have been conducted within the Klamath Basin and at Honey Lake (Miller and Collins 1954, Hunt and Naylor 1955, Rienecker and Anderson 1960, McLandress et al. 1996). These and other wildlife refuges along the Pacific Flyway support relatively large numbers of nesting ducks. However, understanding smaller waterfowl nesting communities may also contribute to the management of duck populations within the Pacific Flyway (Bogiatto 1998). To aid in the formation of future management programs in this region, knowledge of species-specific nest success, as well as patterns of habitat utilization is needed (Maxson and Riggs 1996).

Eagle Lake, Lassen County, is the second largest natural lake in California (Lederer 1976), yet it supports a relatively small nesting duck community (Bogiatto 1998). The paucity of data on over-water nesting ducks at Eagle Lake prompted our study, conducted during the 2000-2001 nesting seasons. The objectives of this study were to: 1) Determine nesting population sizes of over-water nesting ducks at Eagle Lake; 2) Quantify nesting habitat and nest site characteristics; 3) Estimate nest success; and 4) Determine whether nest success is influenced by nest site characteristics such as nest cover density, cover height, percent cover, and water depth.

STUDY AREA

Eagle Lake is located approximately 39 km north of Susanville in Lassen County, California at an elevation of 1570 m (Fig. 1). The Lake is bordered to the southeast, south, and west by forests of Jeffrey pine, *Pinus jeffreyi*, incense-cedar, *Calocedrus decurrens*, and white fir, *Abies concolor*; the Sierra Nevada mountains join the southern Cascades immediately south of the lake. To the east and north lies the arid Modoc Plateau region of the Great Basin dominated by big sage, *Artemisia tridentata*, gray rabbitbrush, *Chrysothamnus nauseosus*, and western juniper, *Juniperus occidentalis*.

Eagle Lake is approximately 22 km long and 4-7 km wide, with a total surface area of approximately 11500 ha. Eagle Lake's three interconnected basins lie on the southwestern corner of the Modoc Plateau; the two northern basins are relatively shallow (6-10 m), while the larger south basin is much deeper (up to 30 m).

The marsh habitats of Eagle Lake are dominated by stands of hardstem bulrush and baltic rush, with small patches of broadleaf cattail. Rush grows primarily where the soil is exposed or in very shallow water.

We selected four study sites based on the presence of nesting habitat and active duck nests: North Basin (NB), Troxel Bay (TR), Duck Island Bay (DI) and Delta Bay (DB) (Fig. 1). NB is located at the north end of the lake. Relatively shallow water, dense submergent vegetation, and dense stands of bulrush characterize this site. The dominant submergents at the NB were sago pondweed, *Potamogeton pectinatus*, and

² Featherstone, J.D. 1975. Aspects of nest site selection in 3 species of ducks. Ph.D. Dissertation. University of Toronto, Ontario. 106pp.

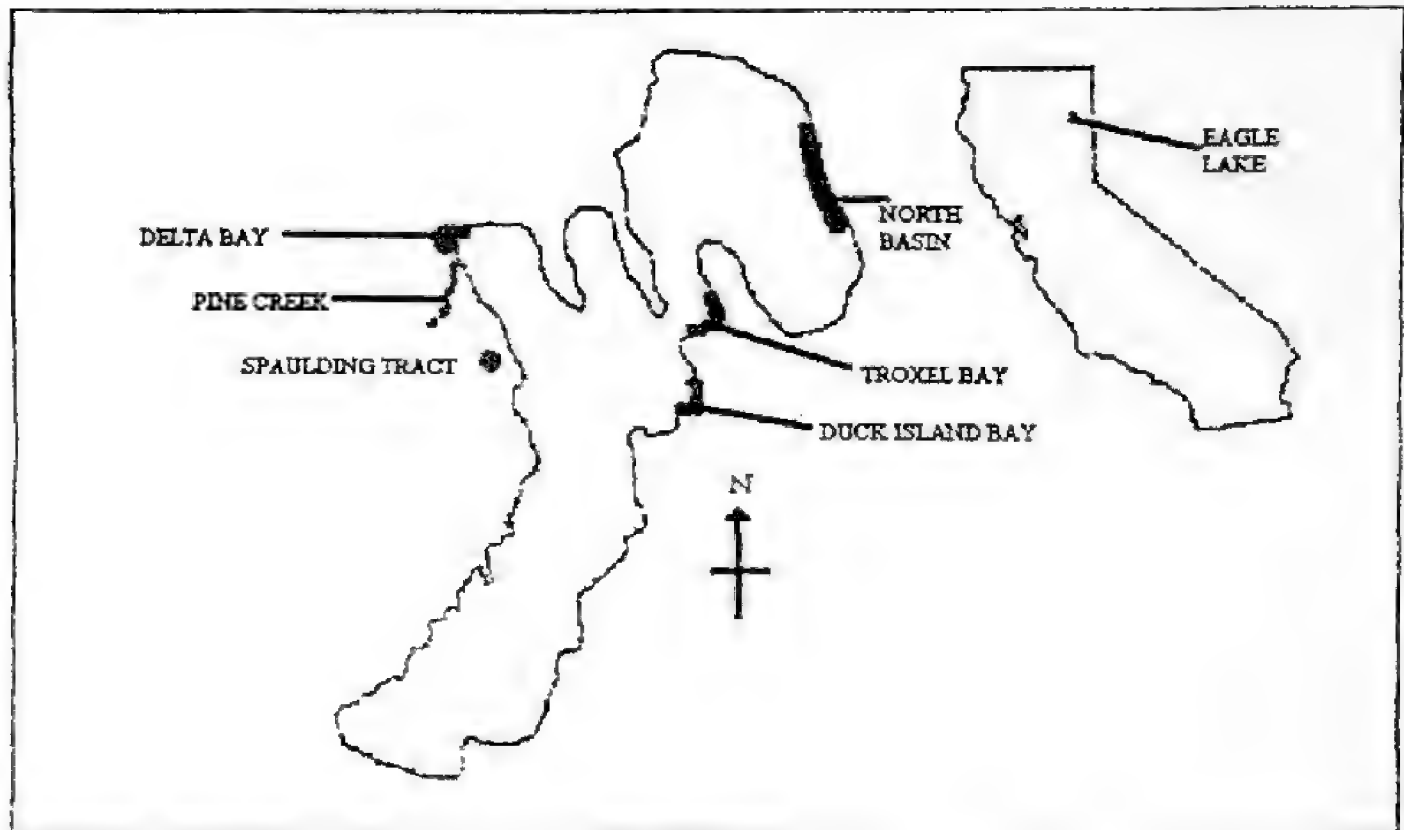


Figure 1. Eagle Lake, Lassen County, California, showing the location of the North Basin (NB), Troxel Bay (TB), Duck Island Bay (DI) and the Delta Bay (DB) study sites. (Adapted from Bogiatto, R.J. 1998. Nesting ecology of ducks at Eagle Lake, Lassen County California. California Fish and Game 84(2): 61-73). Reproduced by permission.

bladderwort, *Utricularia vulgaris*. The larger stands of bulrush at NB are often separated by areas with relatively sparse new growth bulrush, and large open water areas and channels. Areas of emergent vegetation border the water's edge, extending out from shore for up to 300 m, depending upon the water level. Most of the emergents at NB are in shallow water (< 1.5 m), although there is a small area with sparse deep-water vegetation (> 2 m). The shoreline of NB is lined with dense stands of rush.

TR is located to the southeast of NB, on the east side of the Lake. This small cove has two distinct bands of emergent vegetation separated by a body of open water. The first, a pure stand of bulrush, lies in deep water (> 2 m). The second band stretches out from the shoreline (water < 1.5 m deep); with rush growing closest to the shore and bulrush extending out into the Lake. TR is characterized by dense stands of bulrush, interspersed with channels and several small bodies of open water. The TR shoreline is also covered with decaying stands of bulrush. The submergents, primarily sago pondweed and bladderwort, grow in medium to dense stands throughout TR.

DI is located southeast of TR, also on the east side of the Lake. This site is characterized by shallow water (< 1.5 m deep), with several small stands of bulrush extending out into deep-water, as well as several canals, and with a broad range of vegetative densities. Bulrush grows throughout most of the site; with rush only growing along the shoreline. DI has a few sparse stands of submergents, primarily pondweeds.

The fourth site, DB, is located on the west side of the lake, near the mouth of Pine Creek, the Lake's principal tributary. Pine Creek enters the Lake just north of Spaulding, the largest housing tract on the Lake. The emergent vegetation at DB stretches out approximately 10-75 m from shore, with rush growing along the shoreline and bulrush extending out into the Lake; the distribution of vegetation patches varies with ever-changing water levels. Overall the vegetation at this site is sparsely distributed, with intermittent stands of dense bulrush. The submergents, primarily sago pondweed and bladderwort, are widely distributed with several areas of dense growth.

METHODS

Nest Searches

Field surveys were conducted from mid-May through late August or early September 2000 and 2001. Sites were surveyed during 1-2 successive days every 7-14 days. To minimize nest abandonment and maximize attendance of the female, all surveys were conducted between 0700 and 1300 h (Gloutney et al. 1993). To minimize researcher impact, the first visit to a nest was kept to a maximum of 10 minutes; revisits were kept to a maximum of 5 minutes.

A nest was defined as a bowl that contained at least one egg and was in use when found (Livezey 1981). We located nests through systematic searches of all over-water habitats at each of the four study sites; without regard for vegetation type or density. Two or three persons walking 5-10 m apart searched the shallow areas of the marsh; deep-water emergents were searched using a kayak.

All efforts were made to avoid disturbing the surrounding cover when approaching a nest. Once located, nests were marked with white flagging bearing an identification number. Flagging was attached to vegetation approximately 2 m from the nest bowl to facilitate relocation.

Nest Site Characteristics

Nest location, nesting species, water depth, cover species, and height of cover above the nest bowl at the time of initial location were recorded at each site. As females were rarely seen on nests, species was determined by egg size and color (Bellrose 1980, Baicich and Harrison 1997), as well as by breast feathers and down if present (Broley 1950). Water depth and cover height were measured to the nearest 0.025 m. Water depth and height of vegetative cover at successful and failed nests were compared using ANOVA, the null hypothesis being that nest success was independent of water depth and vegetative cover height.

A density cube (Jones 1968, Livezey 1981, Maxson and Riggs 1996), with 100 squares per side, was used to estimate visibility of the nest bowl from the four cardinal directions and from above. The cube was oriented to the cardinal directions and placed in the nest bowl. All squares more than 50% visible from a distance of 1.5 m and at a height of 0.5 m above the surface of the water were counted on each side (Livezey 1981, Maxson and Riggs 1996). The density cube was also used to estimate nest visibility

from above, using photographs taken from 1.5 m above the nest bowl (Featherstone² 1975). For those nests under observation for more than 21 days, visibility measurements were taken twice during each nestling season, to account for new growth. The first measurement was taken upon nest discovery, and the second upon nest completion.

Percent cover at each nest site was estimated using a 0.25 m² quadrat, placed 0.5 m from the nest bowl at each of the four cardinal directions (Maxson and Riggs 1996). As with nest visibility, percent cover was measured twice at sites under observation for more than 21 days.

The null hypothesis that nest success was independent of vegetative density and percent cover was tested using multiple regression analysis for each nesting species.

NEST SUCCESS

Four categories were used in classifying nest fate. The categories are as follows: 1) Successful nests. As hatching is rarely witnessed, a successful nest is identified by the presence of shells and fully separated shell membranes (Rearden 1951); 2) Destroyed nests. Destroyed nests are those unsuccessful nests reaching termination due to some factor other than those initiated by the host; 3) Abandoned nests. Those nests that were abandoned by the female prior to hatch; eggs are generally cold and often decaying; 4) Fate unknown. The fate of some nests could not be determined due to ambiguous evidence or the lack of evidence altogether.

Nest success is defined as the proportion of nests in which at least one egg hatches successfully (Miller and Johnson 1978). Nest success was calculated using both the Apparent (Traditional) Method, whereby the number of successful nests is divided by the total number of nests of known fate, and the Mayfield 40% Method (Mayfield 1961 and 1975, Miller and Johnson 1978). The Mayfield Method takes into account biases introduced due to incomplete nest histories; success is calculated based on exposure days. More specifically the Mayfield 40% Method assumes that failed nests survived through 40% of the last exposure period.

Eggs deposited by interspecific brood parasites (IBP) were identified to species, counted, and measured. Conspecific brood parasitism (CBP) was much more difficult to identify, as host and parasitic eggs are often indistinguishable. Abnormally large clutches (greater than 14 for the redhead and lesser scaup, and greater than 12 for the ruddy duck and cinnamon teal) were attributed to CBP. Predation was classified to species whenever possible using criteria suggested by Rearden (1951).

RESULTS AND DISCUSSION

Nesting Community

The ruddy duck was the most abundant nesting duck at all sites, comprising 47% of all nests located ($n=38$ in 2000, $n=9$ in 2001). Redheads ($n=12$ in 2000, $n=5$ in 2001), lesser scaup ($n=7$ in 2000, $n=2$ in 2001) and cinnamon teal ($n=3$ in both 2000 and 2001) nested in smaller numbers, comprising 17%, 9% and 6% of the nesting community respectively. Nests that remained unclassified due to brood parasitism (BP) and those

that were inactive upon initial location comprised 21% ($n=21$) of all nests located.

Several studies have addressed the importance of emergent vegetation as nesting cover for these and other diving ducks (Low 1941, Rienecker and Anderson 1960, Lokemoen 1966, Featherstone² 1975, Maxson and Riggs 1996). Although some individuals are known to nest in uplands (McKnight 1974), ruddy ducks and redheads are chiefly over-water nesters (Low 1941, Miller and Collins 1954, Hunt and Naylor 1955, Rienecker and Anderson 1960), and thus it is not surprising that these species are the most common over-water nesters at Eagle Lake. Although scaup and teal are known to nest over-water (Miller and Collins 1954, Hunt and Naylor 1955, Rienecker and Anderson 1960), studies in which several habitat types were surveyed indicate that they nest chiefly in uplands (Miller and Collins 1954, Hunt and Naylor 1955, Rienecker and Anderson 1960). Bogiatto (1998) found lesser scaup to be the most abundant upland nesting duck at Eagle Lake. Therefore, it is not surprising that relatively few over-water nests of these two species were found.

Nest Success

Due to the difficulty in distinguishing host from parasitic eggs, those nests incurring CBP were not included in nest success calculations. Additionally, only those interspecifically parasitized nests in which the host species could be determined were included in nest success calculations.

Nest success varied widely between sites and among species. Apparent nest success for ruddy ducks ranged from 0% (DB, 2000 and 2001) to 100% (DI, 2001); Mayfield estimates ranged from 0.3% (DB, 2001) to 31.4% (NB, 2001) (Tables 1 and 2). Estimates of Apparent nest success for redheads ranged from 0% (TR, 2001) to 100% (DB, 2000 and DI, 2001); Mayfield estimates ranged from 0.3% (DI and TR, 2001) to 51.3% (DI, 2000). Estimates of Apparent nest success for scaup ranged from 0% (DB, 2000 and NB, 2001) to 100% (DI, 2001); Mayfield estimates ranged from 1.1% (TR, 2000) to 14.4% (DI, 2000). Cinnamon teal were only found nesting over-water at NB during this study. Apparent and Mayfield estimates show 0% and 5.3% nest success in 2000, and 66.7% and 63.4% in 2001 respectively.

Assuming normal levels of duckling mortality, hen survival, and philopatry, studies from the Prairie Pothole region suggest that Mayfield estimates of 15% nest success for mallards and 20% for late nesting species such as redheads and scaup are needed to sustain a population (Cowardin et al. 1985, Klett et al. 1988). At Eagle Lake, lake-wide Mayfield estimates for the four species ranged from 3.9 to 63.4%, with population sustaining levels of nest success reached by the ruddy (2001), scaup (2001) and cinnamon teal (2001). However, Bogiatto (1998) found lesser scaup to be the most abundant upland nesting duck at Eagle Lake, with cinnamon teal nesting in lower numbers. As Mayfield 40% estimates of nest success was 34% on peninsular habitats at Eagle Lake, it was suggested that these habitats were necessary for nesting duck populations to sustain themselves through time (Bogiatto 1998). No such inference can be made from our study due to low nest numbers.

Abandonment was an important factor limiting the success of duck nests. Of all

Table 1. Apparent nest success estimates for over-water nesting ducks at each of the four study sites on Eagle Lake and lake-wide, 2000-2001.

Nesting species	Total nest numbers ^a	Duck Island Bay (%)	Troxel Bay (%)	Delta Bay (%)	North Basin (%)	Lake-wide (%)
Ruddy duck						
2000	33	60.0	8.3	0	25.0	26.5
2001	8	100.0	—	0	40.0	37.5
Redhead						
2000	10	33.3	—	100.0	50.0	50.0
2001	5	100.0	0	—	33.3	20.0
Lesser scaup						
2000	6	33.3	50.0	0	—	33.3
2001	2	100.0	—	—	0	50.0
Cinnamon teal						
2000	3	—	—	—	0	0
2001	3	—	—	—	66.7	66.7

^aTotal nest numbers includes only those nests used for Apparent nest success calculations. Not included in these calculations are: 10 nests that were inactive when located, 11 unclassified nests, 6 conspecifically parasitized nests (4 ruddy duck and 2 redhead) and 3 nests that were never relocated (2 ruddy and 1 scaup).

Table 2. Mayfield 40% nest success estimates for over-water nesting ducks at each of the four study sites on Eagle Lake and lake-wide, 2000-2001.

Nesting species	Total nest numbers ^a	Total exposure days	Duck Island Bay (%)	Troxel Bay (%)	Delta Bay (%)	North Basin (%)	Lake-wide (%)
Ruddy duck							
2000	33	535	19.8	2.5	3.6	20.2	9.6
2001	8	182	^b	—	0.3	31.4	31.5
Redhead							
2000	10	153	51.3	—	^b	0.7	16.6
2001	5	94	0.3	0.3	—	15.5	5.8
Lesser scaup							
2000	6	96	14.4	1.1	^b	—	3.9
2001	2	61	^b	—	—	5.7	45.0
Cinnamon teal							
2000	3	42	—	—	—	5.3	5.3
2001	3	92	—	—	—	63.4	63.4

^aTotal nest numbers includes only those nests used for Mayfield 40% nest success calculations. Not included in these calculations are: 10 nests that were inactive when located, 11 unclassified nests, 6 conspecifically parasitized nests (4 ruddy duck and 2 redhead) and 3 nests that were never relocated (2 ruddy and 1 scaup).

^b1 redhead, 1 ruddy duck and 2 lesser scaup nests were eliminated from site specific nest success calculations, due to lack of data. Only 1 successful nest was found at these sites, thus Mayfield 40% could not be used to estimate nest success for these sites. These nests were however, used in species-specific lake-wide nest success estimates.

nests located in 2000 and 2001, 44.9% and 28.6% were abandoned, respectively. Redhead and ruddy females are relatively intolerant of continued disturbances and parasitic intrusions, often leading to abandonment by the host female (Weller 1959). Twenty nests (29.3% of all nests located) incurred BP and were later abandoned in 2000 (4 redhead, 6 ruddy, and 10 unclassified); 2 nests (9.5% of all nests located) in 2001 incurred the same fate. As may have been the case in this study, BP was found to be the greatest single factor contributing to nest desertion by Low (1940). Communal or dump nests were much more common in 2000, where 10 nests containing abnormally large clutch sizes were found (the largest of which contained 21 eggs); only 2 dump nests were found in 2001. In all cases, dump nests were abandoned by the host female.

Reasons for abandonment are often difficult to assess. A probable cause could not be identified for 14 (40%) of the deserted nests in 2000 and 1 nest in 2001. Muskrats were observed in the nests of ducks and other species, and nest abandonment often coincided with the accumulation of mounded vegetation in the nest bowl. Miller and Collins (1954) identified the activity of muskrats, *Ondatra zibethica*, as one possible cause of nest desertion. Other potential factors influencing nest abandonment at Eagle Lake include human disturbance and predation on nesting females by raptors such as the great horned owl, *Bubo virginianus*, (Keith 1961, Greenwood et al. 1995, Bogiatto 1998).

No nests at Eagle Lake were lost or abandoned due to flooding; lake water levels rarely rise after the onset of the over-water nesting season. As the summer progresses, lake levels generally drop, often resulting in the disappearance of the shallow water beneath over-water nests. Nests with little or no water beneath them become extremely vulnerable to mammalian predators, such as raccoons and skunks (Stoudt³ 1965, Lokemoen 1966). Of the six nests abandoned in 2001, three were 'dry' nests (two redheads, one ruddy). Lokemoen (1966) reported that redheads usually abandon 'dry' nests, and Featherstone² (1975) found water beneath the nest to be a requirement for ruddies and redheads; nests without water were either destroyed or abandoned.

In addition to nest desertion, predation was also an important factor limiting nest success. Predation was the only observed cause of nest failure at Eagle Lake. Predators destroyed 27.3% of all duck nests located in 2000 and 2001. Evidence of predation during this study includes missing eggs, eggs with holes, broken or crushed eggshells either in the nest bowl or scattered about, and nest bowls that were pulled apart. Although predator identification could not be determined in most circumstances, potential predators in the area include raccoons, striped skunks, *Mephitis mephitis*, mink, *Mustella vison*, California gulls, *Larus californicus*, common ravens, *Corvus corax*, gray fox, *Urocyon cinereoargenteus*, and coyote, *Canis latrans*. The eggshells within several depredated nests showed characteristics which compared positively to those expected from mink predation (Rearden 1951); mink-like scat was found at one such nest. Also, characteristic signs of coyote predation, such as destroyed nest bowls, trampled vegetation and scattered or crushed egg shells (Sooter 1946), were

³ Stoudt, J.H. 1965. Project report on habitat requirements of the canvasback during the breeding season. U.S. Fish and Wildlife Service Project A-8. 6pp.

noted at several depredated nests in 2000. Bogiatto (1998) found raccoons and coyotes to be common predators of upland nests at Eagle Lake.

NESTING COVER AND WATER DEPTH

Hardstem bulrush was the most commonly used nesting cover by over-water nesting ruddies, redheads and scaup. Most studies suggest that bulrush is the preferred nesting cover in over-water habitats when available (Low 1941 and 1945, Hochbaum 1944, Miller and Collins 1954, Weller 1959, Rienecker and Anderson 1960, Featherstone² 1975). Of all nests found at Eagle Lake, bulrush served as a cover plant for 87.2% of ruddy nests, 81.3% of redhead nests, and 31.3% of scaup nests during 2000. In 2000, rush was used as nesting cover by 12.8% of ruddy ducks, 18.8% of redheads, 68.7% of scaup, and 100% of teal. All observed over-water nesting ducks used bulrush as a nest cover plant in 2001, presumably a result of the unavailability of rush due to low lake levels.

Ruddy ducks utilized the widest range of water depths, ranging from 0.05-0.5 m. Nests of redheads were found over water ranging from 0.05-0.85 m in depth; those of scaup were found over water 0.1-0.5 m deep; and teal over water ranging from 0.05-0.125 m in depth.

Water depths beneath ruddy nests varied significantly between 2000 and 2001 (Student's t-test, $P < 0.05$); nests were constructed in shallower water in 2001 (Table 3). Differences in water depth beneath nests of redheads, scaup, and teal were not

Table 3. Mean water depth and vegetation height at the nest bowl of over-water nesting ducks at Eagle Lake, 2000-2001.

Nesting species	Mean water depth (m) ^a	Mean vegetation height (m) ^a
Ruddy duck		
2000	0.34	1.80
2001	0.23 ^b	1.66
Redhead		
2000	0.36	1.53
2001	0.28	1.44
Lesser scaup		
2000	0.21	1.06
2001	0.13	1.43
Cinnamon teal		
2000	0.10	0.95
2001	0.10	1.70 ^b

^aMean water depth and mean height of vegetative cover significantly differed between species, $p = 0.011$ and $p = 0.0001$, respectively.

^bValues differ significantly ($p < 0.05$) between years.

significant among years (Student's *t*-test, $P > 0.05$). Significantly different water depths were utilized by the four species (ANOVA, $P = 0.011$). Ruddies and redheads tended to nest in deeper water than did scaup and teal; with ruddies nesting in shallower water than redheads. These findings are consistent with those of Featherstone² (1975) and Maxson and Riggs (1996). Scaup and teal are primarily upland nesters (Miller and Collins 1954, Hunt and Naylor 1955, Rienecker and Anderson 1960), thus it is not surprising that they nest in shallower water than do ruddies and redheads.

Yearly water depth data at Eagle Lake were combined for the four species; water depth did not differ between successful and failed nests of ruddy ducks, scaup, and teal (ANOVA, all $P > 0.05$). However, in 2001 successful redhead nests were found in significantly deeper water than failed nests (ANOVA, $P < 0.05$). Featherstone² (1975), who suggested that water depth is positively correlated with nest success in ducks, found successful ruddy nests in water approximately 18 cm deeper than unsuccessful nests. The inconsistency of our data for ruddy ducks with those of Featherstone² (1975) may be explained by the relatively shallow water used by over-water nesters at Eagle Lake. Mammalian predators may not be as deterred by the shallow water as they would be by the deeper water noted by Featherstone². Additionally, mammalian predation pressures may be greater at Eagle Lake than those observed by Featherstone² (1975). These conflicting results may also be explained by relatively low nest numbers at Eagle Lake.

The four species of ducks utilized significantly different heights of vegetation as nesting cover (ANOVA, $P = 0.0001$) (Table 3). Ruddies tended to nest in the tallest vegetation, followed by redheads, scaup and teal. These findings are consistent with those of Miller and Collins (1954). Ruddies tend to be late nesters and therefore, the difference in height utilization may be a function of season rather than choice. Teal nested, on average, in relatively tall emergent vegetation during the 2001 nesting season. Shorter stands of rush were not available in 2001, and therefore teal were forced to shift to the taller bulrush stands.

Dwernychuk and Boag (1972) have suggested that nest success is positively correlated with cover height above the nest bowl. Nesting cover height at Eagle Lake did not differ between successful and failed nests (ANOVA, $P > 0.07$). This may be due to differences in predation pressures at the two study lakes. Additionally, our study at Eagle Lake produced a relatively small number of successful nests available for statistical comparison.

Percent Cover and Vegetative Density

During 2000 and 2001, ruddy ducks, redheads, scaup, and teal utilized nest sites with statistically similar (ANOVA, $P > 0.05$) levels of vegetative cover (Table 4). During 2000, these four species were also found nesting in areas with similar vegetative density (ANOVA, $P > 0.05$) (Table 5). In contrast, 2001 data indicate that these species used nest sites with significantly different vegetative densities as viewed from above (ANOVA, $P < 0.05$). Ruddy nests were significantly less visible from above than were those of redhead, scaup and teal; again, this may be due to season rather than choice.

Table 4. Mean percent vegetative cover at successful and unsuccessful over-water duck nests at Eagle Lake, 2000-2001^a

Nesting species	Percent cover at successful nests	Percent cover at unsuccessful nests
Ruddy duck		
2000 (n=33)	43.3	48.2
2001 (n=8)	34.5	38.7
Average ^c	^b	^b
Redhead		
2000 (n=10)	34.5	49.3
2001 (n=5)	7.5	19.7
Average ^c	^b	^b
Lesser scaup		
2000 (n=6)	62.1	52.4
2001 (n=2)	23.8	41.3
Average ^c	42.9	46.9
Cinnamon teal		
2000 (n=3)	—	70.3
2001 (n=2)	60	66.3
Average ^c	—	68.3

^aOnly nests used in nest success calculations were used to calculate mean percent vegetative cover.

^bValues differ significantly between years, thus an average was not calculated (all $P < 0.05$).

^cWhen values did not differ significantly between years ($P > 0.05$), an average was calculated.

Nest cover data for scaup and teal did not differ between years, and did not show significant differences in percent cover or nest visibility from above or the four cardinal directions at successful and unsuccessful nests (multiple regression, all $P > 0.05$). Percent vegetative cover differed significantly (t-test, all $P < 0.05$) between years for ruddies and redheads. Percent cover and nest visibility from above and laterally did not significantly differ between successful and unsuccessful ruddy or redhead nests (multiple regression, all $P > 0.05$).

Previous studies investigating the relationship between nest site characteristics and nest success have yielded mixed results (Odin 1957, Featherstone² 1975, Livezey 1981, Hines and Mitchell 1983, Krasowski and Nudds 1986, Maxson and Riggs 1996). Odin (1957), Featherstone² (1975) and Livezey (1981) concluded that vegetative density is positively correlated with nest success. Nevertheless, our data collected at Eagle Lake are consistent with those of Krasowski and Nudds (1986) and Maxson and Riggs (1996), who found that nest site characteristics did not differ between successful and unsuccessful nests.

Table 5. Mean percent nest visibility from the side and from above at successful and unsuccessful over-water duck nests at Eagle Lake, 2000-2001^a

Nesting species	Lateral View		Overhead	
	Successful	Unsuccessful	Successful	Unsuccessful
Ruddy duck				
2000 (n=33)	14.5	28.2	57.3	70.3
2001 (n=8)	23.6	32.6	67	67.4
Average ^c	19.1	30.4	^b	^b
Redhead				
2000 (n=10)	27.3	22	82.3	86.7
2001 (n=5)	60.8	38.6	100	92.8
Average ^c	^b	^b	91.2	89.7
Lesser scaup				
2000 (n=6)	30.3	12.2	72.8	99.7
2001 (n=2)	12.8	29.6	87	88
Average ^c	21.5	20.9	79.9	93.9
Cinnamon teal				
2000 (n=3)	—	28.7	—	87
2001 (n=2)	13.6	14.1	33	57
Average ^c	—	21.4	^b	^b

^aOnly nests used in nest success calculations were used to calculate mean percent nest visibility.

^bValues differ significantly between years, thus an average was not calculated (all $P < 0.05$).

^cWhen values did not differ significantly between years ($P > 0.05$) an average was calculated.

The conflicting results of the previously mentioned studies may be explained by the hypothesis proposed by Clark and Nudds (1991) stating that the importance of nest concealment on nesting success is dependent upon the makeup of the local predator community. Nest concealment typically is of little importance to nest success in habitats where mammalian predators predominate over avian predators. Avian predators probably play a lesser role at Eagle Lake, as mammalian predators and their signs were frequently sighted during surveys. Nesting cover conceals nests from visually-oriented predators such as birds, but functions only as a physical barrier to scent-oriented predators such as mammals (Joyner 1975, Bouffard et al.⁴ 1988). Bowman and Harris (1980) found no difference in the proportion of partially and totally concealed nests located by raccoons. Additionally, high density vegetation does not provide

⁴ Bouffard, S.H., D.E. Sharp, and C.C. Evans. 1988. Overwater nesting by ducks: a review and management implications. D.W. Uresk, G.L. Schenbeck and R. Cefkin, tech. coords. Eighth Great Plains wildlife damage control workshop proceedings. USDA Forest Service General Technical Report RM 154. p 153-158.

sufficient protection where predator populations are elevated (Stoudt³ 1965, Krasowski and Nudds 1986).

Brood Parasitism

The four species of over-water nesting ducks at Eagle Lake all served as hosts for brood parasites during this study. Parasitic species included the ruddy duck, redhead, and scaup. Of all over-water duck nests found in 2000, 19 (24.1%) incurred BP (Table 6). Of all nests located in 2001, 5 (23.8%) incurred BP. No difference in the rate of BP among the four species (ANOVA, all $P > 0.05$) or between study sites (T-test, all $P > 0.05$) was observed during either year of this study. These findings are consistent with those of Bogiatto (1998) in his study of upland habitats at Eagle Lake. BP by these species is well documented in the literature (Low 1940, Williams and Nelson⁵ 1943, Miller and Collins 1954, Hunt and Naylor 1955, Weller 1959, Rienecker and Anderson 1960, Joyner 1976 and 1983). Nest success, nest visibility, percent cover, water depth, and height of cover did not differ significantly between parasitized and unparasitized nests (multiple regression, all $P > 0.05$).

Table 6. Rate of brood parasitism and success of parasitized over-water duck nests located at Eagle Lake, 2000-2001.

Nesting species	Total nest number ^a	Percent parasitized	Success of parasitized nests (%)
Ruddy duck			
2000	33	13.8	0
2001	8	12.5	100
Redhead			
2000	10	25.0	33.3
2001	5	40.0	0
Lesser scaup			
2000	6	14.2	100
2001	2	0	N/A
Cinnamon teal			
2000	3	0	N/A
2001	3	33.3	0

^aTotal nest number includes only those nests used for BP calculations. Not included in these calculations are: 10 nests that were inactive when located, 11 unclassified nests, 6 conspecifically parasitized nests (4 ruddy duck and 2 redhead) and 3 nests that were never relocated (2 ruddy and 1 scaup).

⁵ Williams, G.S., and M.C. Nelson. 1943. Management of the redhead duck in Utah. Unpubl. Report to U.S. Fish and Wildlife Service. Washington D.C., USA.

Our results suggest that BP does not significantly affect nest success; apparent success of parasitized nests ranged from 0 to 100% (Table 6). However, only those nests active at the time of location and those incurring IBP, facilitating host identification, could be used for nest success calculations. Success of nests incurring CBP could not be determined, due to the similarity in egg morphology between host and parasitic eggs. Therefore, 12 parasitized nests and 8 unclassified nests (20% of total located during 2000-2001) were eliminated from nest success calculations. We therefore suggest that BP (IBP and CBP) may be the single most important factor leading to nest desertion at Eagle Lake; 29.3% of total nests located in 2000 and 9.5% of those located in 2001 were abandoned, possibly due to BP. Although our statistical analysis failed to show negative effects of BP on nest success at Eagle Lake, we strongly suspect that BP and over-water nest success are negatively correlated at Eagle Lake.

MANAGEMENT IMPLICATIONS

Our results are consistent with other studies investigating the importance of emergent vegetation to over-water nesting ducks (Low 1941 and 1944, Hochbaum 1944, Miller and Collins 1954, Weller 1959, Rienecker and Anderson 1960, Featherstone² 1975, Krasowski and Nudds 1986, Maxson and Riggs 1996). These habitat types are often limited at Eagle Lake due to fluctuations in lake levels. During the 2001 nesting season there was a dramatic decrease in the lake level, thereby resulting in less available nesting habitat; a drop in nest numbers was also observed in 2001. During times of fluctuating water levels the relative importance of each stand of emergent vegetation increases significantly.

Our results indicate that predators may play a significant role in waterfowl nest success. Additional studies investigating predator densities at Eagle Lake may be indicated.

Eagle Lake receives a great deal of recreational use by shore and boat anglers. As human disturbance presumably plays a role in nest desertion, an attempt should be made to limit human intrusion into these critical nesting habitats. This may be accomplished by posting signs/bouys around nesting habitat requesting that recreationalists avoid the habitat between May 1 and September 31. Additionally, an attempt should be made to increase public awareness of these nesting habitats and the multitude of species that utilize them. This may be accomplished by posting interpretive signs/displays/fliers or brochures at Eagle Lake marinas, campgrounds, and bait shops.

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FEEDING PREFERENCES AND SIZE-RELATED DIETARY SHIFTS OF TREEFISH (SCORPAENIDAE: *SEBASTES SERRICEPS*) OFF SOUTHERN CALIFORNIA

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ABSTRACT

An analysis of the stomach contents of 74 specimens of treefish, *Sebastes serriceps*, collected from the Southern California Bight revealed that teleosts, crabs, and carideans were the most important prey in the diet of this species. The species of prey largely represented members of the benthic or epibenthic community in rock reefs and kelp forests. The dietary composition of treefish changed with fish size. The diets of smaller fish, 50-199 mm TL, consisted primarily of peracarideans, carideans, and crabs. A dietary shift occurred in fish larger than 200 mm TL, as larger fish showed increasing consumption of teleost fishes, and decreasing consumption of peracarideans and carideans. Based on treefish stomach contents, it appears that *S. serriceps* is a nocturnal benthic forager.

INTRODUCTION

Treefish, *Sebastes serriceps*, are demersal residents of rocky reefs ranging from San Francisco to central Baja California, although they are most common south of Pt. Conception (Miller and Lea 1972, Love et al. 2002). Treefish inhabit caves and crevices during the day, and they may be crepuscular or nocturnal (Hobson et al. 1981). They are found from subtidal waters to depths of 97 meters (Love et al. 2002). They appear to live at least 25 years and can reach 41 cm total length (TL) (Phillips 1957, Colton² 2005). *S. serriceps* has been commercially important since 1875 (Phillips 1957) and contributes to the recreational fishery and the commercial live rockfish fishery (Larson and Wilson-Vandenberg 2001, Love et al. 2002). However, little is known about their general ecology, including population levels, behavior, or diet.

In order to make accurate fisheries management decisions, it is important to determine how the removal of a fished species may impact the ecosystems in which they live (Pilling et al. 2001). By studying feeding habits, inferences about predator behavior

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²Colton, M.A. 2005. Rarity in nearshore fish communities: a study of treefish (Scorpaenidae: *Sebastes serriceps*). Master's Thesis, San Francisco State University, San Francisco. 130 pp.

can be made based on the location and behavior of the prey. For example, Lowe *et al.* (1996) examined stomach contents of tiger sharks, *Galeocerdo cuvier*, and concluded that younger sharks fed nocturnally on benthic organisms, while older sharks fed on the surface during the day. Correlations between feeding behavior and prey items have also been observed in four different species of rockfish (Larson³ 1972). Diets may also change with ontogeny, as observed in copper rockfish, *S. caurinus*, and quillback rockfish, *S. maliger*, by Murie (1995), indicating changing ecological roles for species during their lives.

Quast (1968) observed that treefish diet consisted primarily of shrimp and crabs and other small crustaceans, along with a small number of fish. However, this analysis was based on a small sample size (21 fish, over 80% of which had empty stomachs), and utilized only percent frequency of occurrence as a measure of dietary importance. Hobson *et al.* (1981) examined the stomach contents of 28 treefish, and found shrimp, crabs, and the teleost *Chromis punctipinnis*, which occurred in 7 stomachs.

The purpose of this study is to conduct a more extensive examination of the diet of treefish in order to determine its ecological role in the community, to determine if diet changes with body size, and to make inferences about the feeding behavior of treefish based on its prey choice.

METHODS

A total of 74 treefish was speared by SCUBA divers using pole spears at Santa Cruz, Catalina, and Anacapa islands, between May 2003 and December 2004. Nearly all specimens were captured during daylight hours. Specimens were placed on ice immediately after capture, and then frozen. After thawing, total length and body weight were recorded for each specimen. Stomachs were removed and placed in ethanol. Stomachs were stored in ethanol for at least 1 day before examination. Contents were weighed, counted, separated into prey groups, and identified to the lowest taxonomic level possible using a combination of keys (Clothier 1950, Smith and Carlton 1975, and Morris *et al.* 1980).

A variety of prey was observed in the stomachs, and these were grouped into several categories (Table 1). Crabs were defined as both brachyurans and crab-like anomurans. For comparison among size classes, fish were divided into five size categories: 50-199 mm ($n=9$), 200-249 mm ($n=17$), 250-299 mm ($n=21$), and 300-349 mm ($n=9$). Importance of prey was measured by the percent frequency of occurrence (number of treefish stomachs containing a certain type of prey divided by the total number of treefish stomachs), numerical abundance (number individuals of particular type of prey seen in all stomachs in a sample divided by the total number of all prey), the percent by weight (total weight of a particular type of prey divided by the total weight of all prey), and the Index of Relative Importance (IRI). The IRI was calculated according to Pinkas *et al.* (1971) as:

³Larson, R. J. 1972. Food habits of four kelp-bed rockfishes (Scorpaenidae, *Sebastes*) off Santa Barbara, California. Master's Thesis, U. C. Santa Barbara. 57 pp.

$$IRI = F(N+W)$$

Where: N= Percent numerical abundance

W= Percent by weight

F= Percent frequency of occurrence

IRI= Index of relative importance

These measures were computed for the entire sample and for each size class of treefish.

Percent frequency of occurrence of prey items among length classes were compared using a Chi-square test (SPSS, ver. 12.0). Fish were divided into small (50-199 mm, N=11), medium (200-249 mm, N=28), and large (250-350 mm, N=35) categories, to facilitate analysis.

RESULTS

Of the 74 treefish stomachs examined, 56 (75.7%) contained prey items. Prey categories occurring the most frequently and in the highest numbers in the pooled sample were crabs, carideans, and, to a lesser degree, teleosts (Table 1). Other prey, such as cephalopods, peracaridans, and other phyla were much less common. By weight, teleosts were the predominant contributors to the diet (Table 2). The index of relative importance (IRI) ranked teleosts as the most important prey item (29.60), followed by carideans (22.71), and crabs (15.80). All other prey groups had IRI values of less than 1, with the exception of Cephalopoda at 1.13 (Table 2).

With the exception of squid, *Loligo opalescens*, all of the species of invertebrate prey found (Table 1) were benthic, or possibly associated with algae. The fishes consumed were all residents of the kelp-rock assemblage, and included both strictly benthic species (*Gobiesox*, *Oxylebius*, and Cottidae), epibenthic or canopy species (*Oxyjulis*, *Girella*, and *Medialuna*), and *Chromis*, which occupies both benthic and canopy habitats (Table 1). Within the major prey categories, certain species of prey were often relatively common, such as the crab *Paraxanthias taylori*, the teleost *Chromis punctipinnis*, and Carideans of the genus *Alpheus* (Table 1).

The diet of treefish changed substantially with body size. Small treefish (50-199 mm) ate significantly more peracaridans than medium or large treefish ($X^2=8.645$, $df=2$, $p=0.013$). Large treefish (250-349 mm) ate more teleosts than the two smaller size classes ($X^2=5.948$, $df=2$, $p=0.051$). Carideans and crabs were equally likely to be eaten by all size classes ($X^2=0.327$, $df=2$, $p=0.849$ and $X^2=0.612$, $df=2$, $p=0.736$, respectively). Although we did not divide the smallest size class into smaller length categories because of the small sample size of 100-199 mm fish, the smallest fish (50-99 mm) ate relatively few crabs, suggesting another shift in diet from carideans and peracarideans to crabs in fish greater than 100 mm. Cephalopods and the other category occurred in such low numbers that the Chi-square analysis could not be performed with confidence.

Carideans, peracarideans, and crabs dominated the diet of 50-199 mm treefish, while teleosts absent (Fig. 1A). Teleosts and cephalopods appeared in the diets of 200-249 mm treefish, while crabs and carideans remained important (Fig. 1B). Teleosts had the

Table 1: Numerical composition of the diets in 56 *S. serriceps* stomachs containing prey, showing the categories into which prey taxa were grouped.

Prey group	Prey Identification	N
Crabs	Brachyura:	
	<i>Loxorhynchus crispatus</i>	1
	<i>Paraxanthias taylori</i>	12
	<i>Scyra acutifrons</i>	5
	<i>Cancer antennarius</i>	2
	Unidentified	5
	Anomura:	
	<i>Petrolisthes manimaculis</i>	1
	Unidentified	3
	Total:	29
Caridea	<i>Alpheus</i> sp.	6
	<i>Lysmata californica</i>	1
	<i>Heptacarpus</i> sp.	4
	Unidentified	17
	Total:	28
Peracarida	Isopoda	1
	Amphipoda	
	Gammaridea	2
	Caprellidea	3
	<i>Metacaprella kenmerlyi</i>	1
	Total:	7
Teleostei	<i>Sebastes</i> sp.	1
	<i>Chromis punctipinnus</i>	4
	<i>Oxyjulis californica</i>	1
	<i>Gobiesox rhessodon</i>	1
	<i>Oxylebius pictus</i>	1
	<i>Girella nigricans</i>	1
	<i>Medialuna californiensis</i>	1
	Family Cottidae	3
	Unidentified	8
	Total:	21
Cephalopoda	<i>Loligo opalescens</i>	1
	<i>Octopus rubescens</i>	1
	Unidentified	1
	Total:	3
Other	Echinodermata	
	Class Ophiuroidea	1
	Cirripedia	1
	Bivalva	
	Family Mytilidea	1
	Total:	3

Table 2: Percent frequency of occurrence, percentage numerical abundance, percentage wet weight of prey, and index of relative importance (IRI) for categories of prey in *S. serriceps*. (Note: As some individuals' stomachs contained more than one prey category, the percent frequency of occurrence is greater than 100%.) N = 56 stomachs.

Prey group	Occurrence(%)	Numbers (%)	Wet weight (%)	IRI
Crabs	48.21	24.62	8.16	15.80
Caridea	48.21	42.31	4.79	22.71
Peracarida	8.93	9.23	0.04	0.83
Cephalopoda	5.36	3.08	18.06	1.13
Teleostei	33.93	18.46	68.79	29.60
Other	2.86	2.31	0.15	0.09

highest IRI value of all prey in 250-299 mm treefish, while crabs and carideans remained important (Fig. 1C). Teleosts had by far the largest IRI score among dietary items in 300-349 mm treefish, partly because of their very large relative contribution in weight consumed (Fig. 1D). However, teleosts also occurred in half of the stomachs, at least equivalent to crabs and carideans, which also had relatively high IRI scores. Thus the main ontogenetic change in diet in treefish was the increasing importance of teleosts in the larger size classes.

DISCUSSION

The treefish appears to be a fairly selective forager. The frequent occurrence of the species of brachyura and anomura, of caridans in the genus *Alpheus*, and of cottids and *Oxylebius pictus* in treefish stomachs indicates that this species forages on the benthos. *Chromis punctipinnis* (blacksmith) was also eaten in high numbers, which is out of character for a benthic feeder until *C. punctipinnis*' life history is taken into account. During the day, *C. punctipinnis* can be found midwater; at night they move into benthic caves and crevasses (Ebeling and Bray 1976, Bray 1981). This would indicate that treefish hunts for *C. punctipinnis* at night, or at dusk or dawn, as suggested by (Hobson *et al.* 1981). Nocturnal or crepuscular feeding is also probably common in other benthic rockfishes from shallow rock reefs and kelp forests (Hobson *et al.* 1981, Murie 1995, and R. Larson unpublished data).

Decapods and other benthic dwelling organisms also played an important role in the diet of treefish. *Paraxanthias taylori*, the crab eaten most frequently, can be found in high numbers on the seafloor beneath kelp forests (Morris *et al.* 1980), and was also reported in the diet of treefish by Hobson *et al.* (1981). The importance of *Alpheus* sp. and spider crabs correlates with the findings of Quast (1968) regarding treefish diets. However, contrary to Quast (1968), we found no evidence of stomatopods and polychaetes. This may be because polychaetes are digested faster than crabs and shrimp, or that they were over-represented in Quast (1968) due to the low sample size.

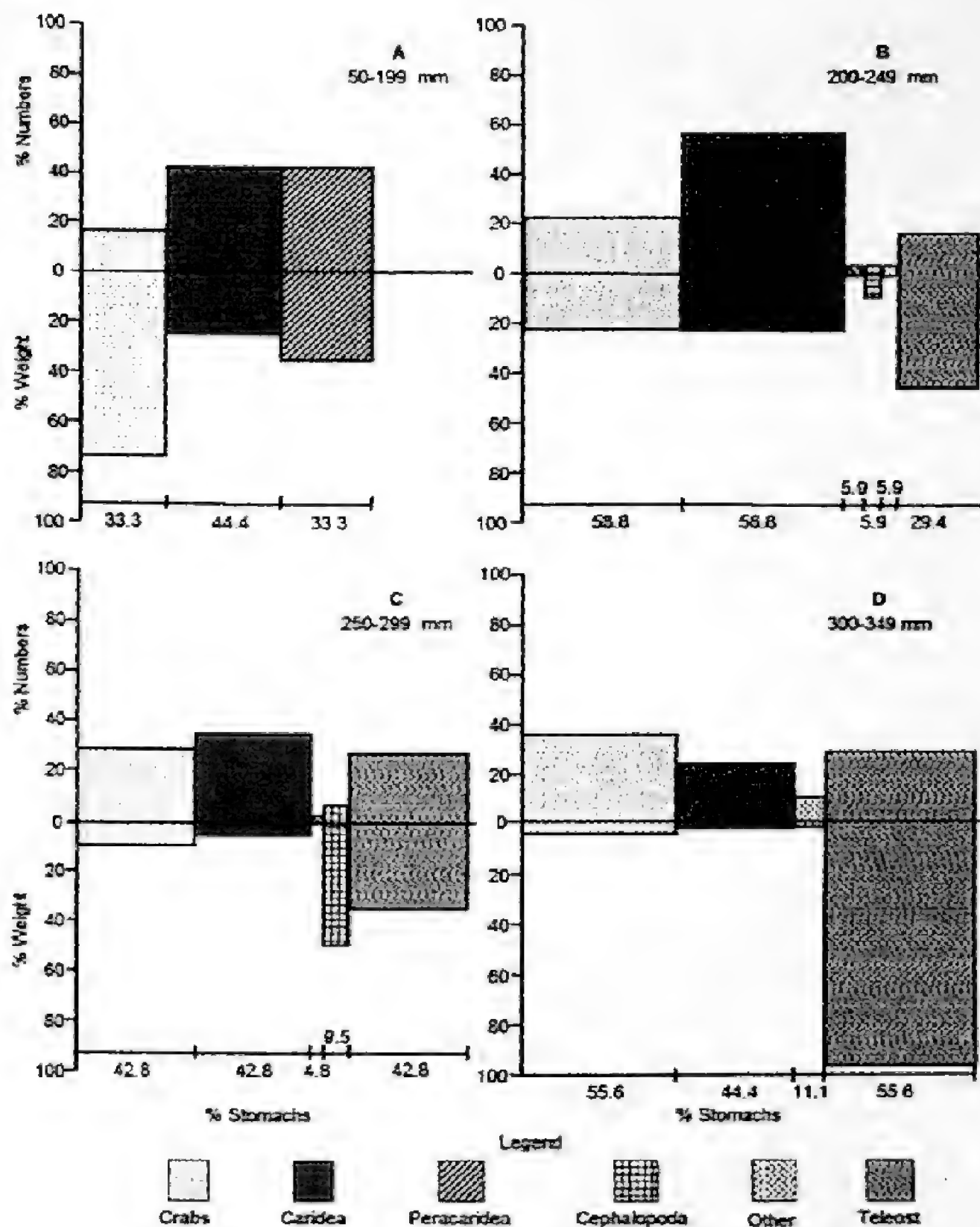


Figure 1: Index of relative importance diagrams for size classes of *S. semiceps*. Sample sizes for the size classes were 9 for 50-199 mm TL (A), 17 for 200-249 mm TL (B), 21 for 250-299 mm TL (C), and 9 for 300-349 mm TL (D).

The correlation of an increase in size with the increasing importance of teleosts as a prey item has been shown in several predatory fishes. The changing importance of particular prey items as a fish grows larger was also illustrated in a study done by Pedersen and Riget (1993) on redfish, *Sebastes* spp., and Greenland halibut, *Reinhardtius hippoglossoides*. Pedersen and Riget (1993) found that larger prey items were found in the guts of larger individuals. Red snapper, *Lutjanus campechanus*, is another predatory fish that experiences a dietary shift at >60 mm (Szedlmayer and Lee 2004). Larger red snappers eat teleosts and greater amounts of crab, while smaller fish eat smaller prey, such as copepods and shrimp. In an interesting study that looked at settlement size of reef fish, *Lutjanus quinquelineatus* was found to settle at a larger size than other reef fish, enabling it to eat larger prey such as fish immediately after settlement (Sweatman 1993), as opposed to having a dietary shift at a later time, as happened in this study. It is likely that the correlation between bigger predators and larger prey in these studies has to do with the gape size of the predators; smaller fish would be incapable of fitting a larger fish through their mouth. Other morphological features, such as a longer digestive tract or a larger number of cecae would allow treefish to digest fish more readily than its conspecifics, as was found in Murie's studies (1995). The treefish appears to forage in similar habitats throughout its life, feeding on organisms found in the benthos, as indicated by predominance of benthic invertebrates and fish in its diet. We also noticed that the small juveniles less than 100 mm TL also consumed benthic invertebrates such as amphipods and carideans. This supports Singer's (1985) theory that the feeding behaviors of juvenile rockfish mirror those of the adults.

This study tried to take into account the relative importance of certain prey items in the diet of treefish using the Index of Relative Importance equation developed by Pinkas (1971); however, there are obvious flaws in this method. Weight was variable depending on the rate of digestion. Shrimps were probably digested faster than crabs, and teleosts were probably digested faster than all other prey groups with the exception of cephalopods (Hopkins and Larson 1991, Singh-Renton et al. 1996). As a result, the already large relative contribution of teleost to the weight of food consumed is probably even greater than shown in our results. In addition, the numbers of small, quickly-digested prey such as carideans and peracarideans may be slightly underrepresented, because of their more rapid turnover in the stomachs of treefish.

The size of the meal along with the type of prey item limited what a fish could consume. For example, large teleosts could not be eaten in the same numbers that smaller crabs could, which skews the numerical abundance count. Size of the prey and rate of digestion are related, which this study did not take into account. In order to truly represent the diets of treefish, digestive rates should be coupled with models of gastric evacuation to compute daily rations for each type of prey.

Based on the data in this study, it appears that the treefish is an important predator in rock-reef habitats off southern California. Other benthic rockfishes, such as the black and yellow rockfish, *S. chrysomelas*, and gopher rockfish, *S. carnatus*, are now uncommon off southern California, and the treefish seems to fill their role as a predator of benthic invertebrates (Larson² 1972). Large treefish also seem to be important

predators of resident rock-reef and kelp-forest fishes. Hallacher and Roberts (1985) showed that some species of rockfish may be important predators on juvenile fishes in kelp forests, and Murie (1995) showed that *S. caurinus* consumed non-resident pelagic fishes. However, our study shows that large individuals of treefish may play an important role as a predator on subadult and adult fishes that reside in kelp forests, since many of the fish observed in the stomachs of treefish were larger than young of the year or small juveniles. While treefish may not be particularly abundant (Colton 2005), large individuals of this species may play an important role in the rock-reef and kelp-forest ecosystem of southern California, and this effect should be taken into account in the management of fisheries for this species (Pilling et al. 2001).

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CONSIDERATION OF THE COMMON NAME FOR *GILA (SIPHATELES) BICOLOR VACCACEPS*, A TUI CHUB IN THE COW HEAD BASIN OF NORTHEASTERN CALIFORNIA

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I propose Cow Head tui chub as the appropriate common name for *Gila (Siphateles) bicolor vaccaceps* (a.k.a. Cowhead Lake tui chub), a small minnow restricted to the Cow Head Basin in extreme northeastern California and northwestern Nevada. The species name *vaccaceps* refers to the geographical locality and not to the character of the fish's head (Bills and Bond 1980). Therefore, separation of the words following geographical convention is appropriate, while elimination of the term "Lake" simplifies the common name without loss of information.

There has been historical confusion with the proper spelling of Cow Head Lake. In accepted geographic usage, the name appears as two separate words (i.e., Cow Head) on most official maps of the area, starting in 1892 (USGS^{1,2,3} 1892, 1993, 2006), although it has also been spelled as "Cowhead" in some geographic literature (Pease 1965, Gudde 1969, Bright 1998). Prior to that the lake was referred to as Pelican Lake (Minto⁴ 1879). In 1964, the U.S. Board on Geographic Names formally designated the official name and spelling as Cow Head Lake (USBGN 1964, Durham 1998), and it appears as such on current topographic maps (USGS² 1993). The U.S. Board on Geographic Names was created in 1890 and is the federal body authorized to establish and maintain uniform geographic name usage throughout the federal government (USGS³ 2006). In the biological literature, the lake has generally been spelled as a single word (i.e., Cowhead; Hubbs and Miller 1948, Hubbs et al. 1979, Bills and Bond 1980, Moyle 1976, 2002). No reason for this spelling has been provided, and it is notable that none of the authors were particularly familiar with the basin, having only passed through for short 1-day visits. Bills and Bond (1980) used the combined form for both the geographical location and common name in their original description of the subspecies (i.e., Cowhead Lake

¹U.S. Geological Survey. 1892. Alturas Sheet, California. 1:250,000 Series (Topographic Map).

²U.S. Geological Survey. 1993. Lake Annie Quadrangle, California-Oregon. 41120-III. 7.5 Minute Series (Topographic Map).

³U.S. Geological Survey. 2006. Cow Head Lake - feature detail report. Geographic Names Information System (GNIS). Reston, Virginia: U.S. Federal Government. The nation's official geographic names repository, 1976 - present. <http://geonames.usgs.gov/> Accessed 22 February 2006.

⁴Minto, W. 1879. Field notes of the division and subdivision lines of Township 47 North, Range 16 and 17 East, Mount Diablo Meridian, California. General Land Office Survey. Washington, D.C., USA.

tui chub), suggesting that they were unfamiliar with the correct geographical spelling. Recent authors have followed the accepted geographic name for the lake and other similarly named features within the Cow Head Basin, but have maintained the original spelling of the fish's common name in the absence of an established alternative, further confusing the reader (Scoppettone and Rissler 2006). Usage of a common name in agreement with the geographical place name would eliminate this confusion.

Elimination of the generic term "lake" follows American Fisheries Society convention for simplicity in common names and is also more descriptive of the taxon's distribution and ecological habit (Nelson et al. 2004). Although the Cow Head tui chub would have utilized the shallow lake, when available, it was clearly not limited to lake habitat. The type locality is a spring and small, vegetation-choked reservoir pool (max. depth 1 m) on the Arrowhead Ranch about 100 m from the northwest shore of Cow Head Lake (Miller⁵ 1939; UMMZ⁶ 136870), and no collections were ever actually made from the lake, which has not been perennially full since the 1920's, due to natural droughts and drainage for agricultural use of the lakebed. Current populations generally occupy shallow spring-fed streams (Keno, Barrel, West Barrel, and Cow Head Slough), and even the Schadler Ditch population is fed primarily by stream flow in the summer (Scoppettone and Rissler 2006). The structure of the Cow Head tui chub's gill rakers is also more characteristic of stream or shallow-water populations. True pelagic lake-dwelling tui chub populations (e.g., Lahontan Lake tui chub, *G. (S.) b. pectinifer*) have longer, slenderer gill rakers, presumably as an adaptation for feeding on pelagic zooplankton, while the Cow Head tui chub's gill rakers are short and stubby, as in other stream or shallow-water populations (Bills and Bond 1980, Moyle 2002).

Adoption of the proposed common name, Cow Head tui chub, will bring the taxon epithet into agreement with the geographic location on which it is based, avoid spelling confusion between the fish and the location, and shorten the name, while better characterizing both the range and habitat of the taxon.

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FIRST RECORD OF THE WOLF-EEL, *ANARRHICHTHYS OCELLATUS* (PISCES: ANARRHICHTHIDAE), FROM BAJA CALIFORNIA, MEXICO

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The wolf-eel (*Anarrhichthys ocellatus* Ayres, 1855) is an elongate, benthic perciform that inhabits the temperate North Pacific from the southeastern Bering Sea and eastern Aleutian Islands to southern California (Mecklenburg et al. 2002) and occurs from the intertidal zone (Miller and Lea 1972) to the offshore shelf, up to 244 m depth (CAS 19447; see Table 1 for abbreviations). Records from the Sea of Japan (Miller and Lea 1972; Eschmeyer and Herald 1983) are not valid and were most likely based on *Anarrhichas orientalis* Pallas, 1814. The species is one of the early fishes described from California (Ayres 1855a and b) and was characterized as, "In color perhaps no more beautiful fish than this has yet been found in our waters." The original two specimens were taken from "the Bay of San Francisco." The southern limit for the species is off Imperial Beach, San Diego County, but it is not common south of Point Conception (Fitch and Lavenberg 1971). Previously, Barnhart (1936) noted it to be "one of our most remarkable fishes: feeding chiefly on sea-urchins and sand dollars" and gave Redondo as the southern limit; Hubbs and Barnhart (1944) extended this to La Jolla Submarine Canyon, San Diego County.

On 26 November 2005, a wolf-eel was photographed by Sean Dyer while SCUBA diving off the Pacific coast of Baja California, Mexico, on Escondido Reef near La Bufadora (approx. lat. 31°42'N, long. 116°42'W) (Fig 1). The photographs represent the first record of a wolf-eel in Baja Californian waters and a new southern limit for the species. This locality is approximately 100 km southeast of Imperial Beach, California.

The wolf-eel was photographed at a depth of 70 ft (21.3 m) at 1100 hr with an Olympus C-5060 Wide Zoom digital camera off the dive boat "Dale's Panga." The water temperature at the bottom (site of observation) was 12.8° C.

Identification was confirmed by the large rounded head and snout, large upturned mouth, and purple-gray color with numerous round spots around the eye and posteriorly on the head and body. The head was estimated as ca. 10 cm in depth which

Table 1. Museum records of wolf-eels from southern California. Listing is by latitude from Point Conception south to San Diego County; two Channel Island records are included.

Locality	Date of collection	Museum number ^a	Depth ^b (in feet)	Size ^c (mm SL)	Method of capture
Off Point Conception	Aug 1974	SIO 74-122	20-60	440	spear
Santa Barbara	1882	USNM 31238	unknown	unknown	unknown (skeleton)
San Miguel Island, Castle Rock	06 Oct 1962	LACM 3334	unknown	unknown	spear
Santa Cruz Island	Mar 1966	LACM 6912-1	unknown	982	hook and line
Harrison's Reef, County Line	23 May 1965	UCLA W65-26	45	455	ichthyocide
Redondo Beach barge reef	29 Dec 1967	LACM 38325-1	90	572	by hand
Off San Pedro	05 Jun 1956	LACM 50122-1	unknown	476	unknown
Horseshoe Kelp, off Long Beach	21 Mar 1961	LACM 51467-1	unknown	820	hook and line
Off La Jolla	03 Apr 1946	CAS 19447 ^d	800	949	setline
Off La Jolla	01 Feb 1953	SIO 53-13	65-70	925	hook and line
Off La Jolla	10 July 1976	SIO 76-269	420-480	2015	gill net
Point Loma	17 Dec 1975	SIO 75-577	50	363	by hand
Off San Diego Bay	28 July 1953	SIO 53-124	unknown	1134	hook and line
4 mi. SW of Pt. Loma	4 Mar 1949	UCLA W49-335	unknown	617	lobster trap

^aCAS – California Academy of Sciences. LACM – Natural History Museum of Los Angeles County. SIO – Scripps Institution of Oceanography. UCLA – University of California at Los Angeles. USNM – United States Museum of Natural History.

^bDepth of capture is given in feet as originally reported.

^cLength is given as Standard Length.

^dCAS 19447. This specimen was originally catalogued as SIO H46-36.

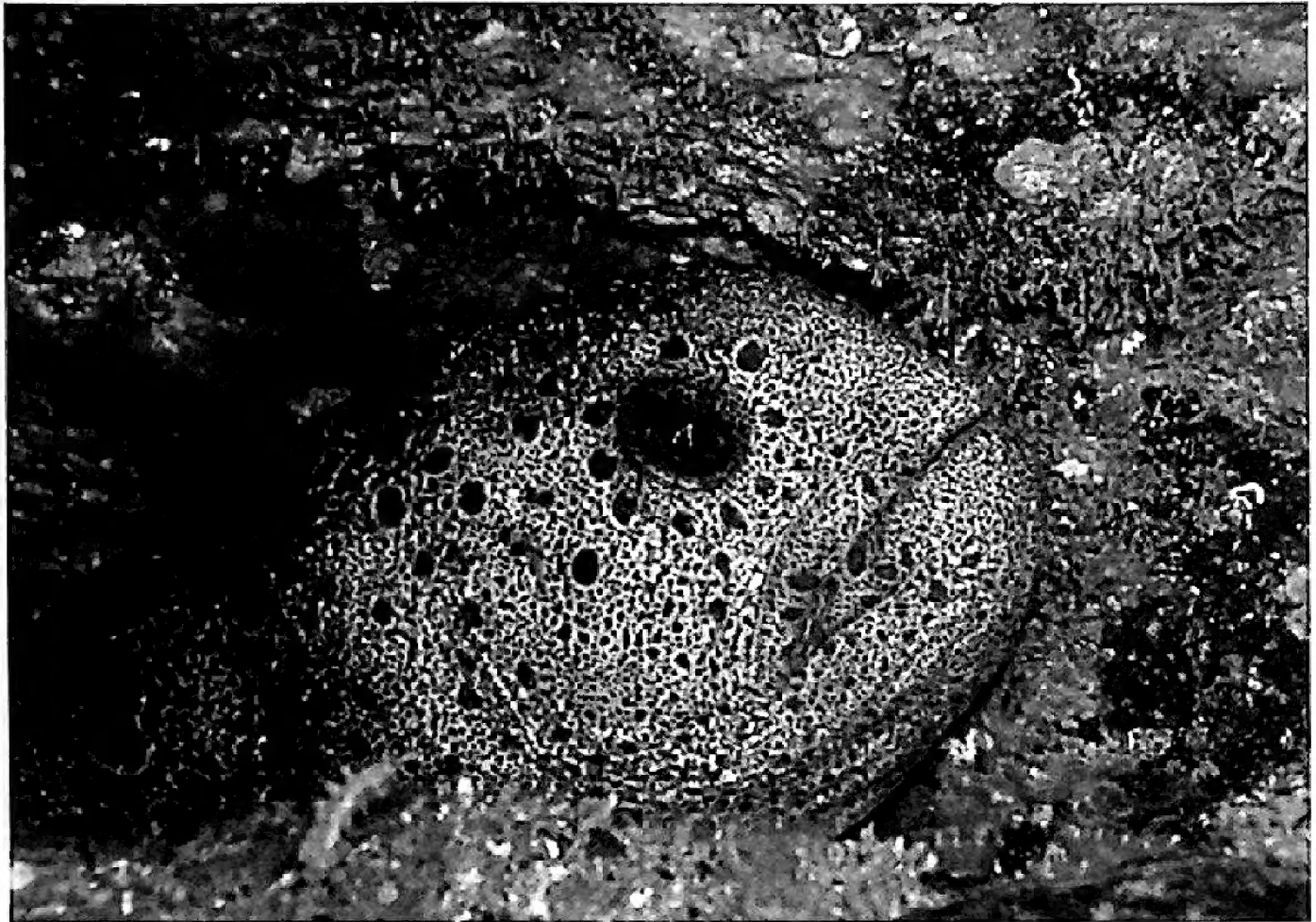


Figure 1. Wolf-eel from Escondido Reef, near La Bufadora, Baja California, Mexico. Photograph by S. Dyer.

would make the wolf-eel roughly 125 cm in length, based on known body proportions. Three digital photographs were taken; two are lateral views of the head protruding from a crevice and the third is an anterior view showing more of the body wrapped around in the crevice. Copies of the photographs have been cataloged and stored in the Ichthyology files at the Natural History Museum of Los Angeles County (LACM 56373-1).

Fourteen museum records of wolf-eel are known from southern California of which six are from San Diego County (Table 1). The previous southern record from Imperial Beach is not supported by a museum specimen but is listed in Radovich (1967, Table 6): "Wolf-eel - 1 - May 22, 1958 - 1 to 1½ miles off Imperial Beach, San Diego County, in 8½ fathoms." The nearest museum specimen was collected in 1949 and housed at UCLA (W49-335) from "4 mi. SW of Point Loma", estimated to be just north of Imperial Beach.

The Pacific coast of northern Baja California is known to be an area of strong upwelling (Bakun and Nelson 1977), bringing cold water close to shore and creating a cold-temperate environment for populations of shorefishes usually more common north of Point Conception (Stepien et al. 1991; Robinson 2004). This record represents another cold-temperate species occurring in the cool waters off northern Baja California.

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